Life cycle assessment of shredder residue management

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Preface

This report provides a life-cycle assessment (LCA) of the treatment of shredder residue (SR) in Denmark.

The LCA was conducted for the Environmental Protection Agency by DTU Environment in the period March-July 2014, as part of a service agreement between the Danish Environmental Protection Agency and the Technical University of Denmark on research-based services in the field of waste management. The report is part of a larger survey on improved resource recovery of waste, focusing on the environmental as well as socio-economic consequences of different treatment scenarios for shredder waste, impregnated wood waste, wood waste for recycling and district heating pipes. The LCA was conducted using the EASETECH LCA model developed by DTU Environment for the environmental assessment of waste management systems and environmental technologies.

The LCA was conducted in accordance with the LCA principles outlined in DS/EN ISO standards 14040 and 14044. A critical review was carried out by external LCA experts from the Danish Technological Institute. A reference group consisting of Danish stakeholders with interests in SR management were asked to comment on the report as well. All critical comments from reference group and LCA reviewer were followed and the report was changed accordingly.

The Danish stakeholders were represented by the Innovation Partnership for Shredder Waste, whose head of secretariat is Jette Bjerre Hansen, DAKOFA.

The Danish Environmental Protection Agency was represented by Trine Leth Kølby and Thilde Fruergaard Astrup.

The report was prepared by Alessio Boldrin, Line Kai-Sørensen Brogaard, Anders Damgaard and Thomas Astrup from DTU Environment.

DTU 2014

Summary and conclusion

Background
Shredder residue (SR) is the "residual fraction from mechanical shredding of metal containing scrap originating from different sources and processed at recycling stations or metal recovery businesses" (Hyks et al., 2014). Danish SR is typically a mixture of 20% automotive shredder residues (ASR) and 80% of other residues originating from different sources (e.g. white goods, dismantled boats, industrial bulky waste, metal scrap). SR is currently disposed of in hazardous waste landfills in Denmark; however, legislative and economic issues mandate exploring alternative solutions for its management.

Introduction
The present report was prepared by DTU Environment and includes a life cycle assessment (LCA) of the treatment of SR in Denmark. The assessment compares the potential environmental impacts and depletion of abiotic resources in relation to four alternative scenarios, all including the sorting of recyclables and the management of residual material through thermal treatment and/or disposal in controlled landfills after biological stabilisation.

Method
The consequential LCA was carried out according to ISO standards 14040 and 14044, while the choice of life cycle impact assessment (LCIA) methods and impact categories followed the recommendations set out in the International Reference Life Cycle Data System ILCD handbook 'Recommendations for Life Cycle Impact Assessment in the European context' (JRC, 2011). System expansion was used to solve process multi-functionality and to credit production avoided through reuse, recycling and recovery. The LCA was carried out using the EASETECH model developed at the Technical University of Denmark. Uncertainty calculations were carried out via Monte Carlo simulation, using 10,000 random variables to calculate the uncertainty distribution of the results.

Scenarios
Four scenarios for the treatment of SR were analysed. All scenarios included the sorting of recyclables (i.e. glass, plastic and metals) and the landfilling of fine residues <4 mm, which were assumed not suitable for thermal treatment because of their high content of heavy metals. Individual scenarios then included alternative treatment of the >4 mm residue remaining after sorting of recyclable materials: landfilling in Scenario 1 (S1), co-combustion at a waste incineration plant in Scenario 2 (S2), pyrolysis in Scenario 3 (S3), and co-combustion in a cement kiln in Scenario 4 (S4).

Inventory data
Inventory datasets regarding SR composition and treatment technologies were developed mostly based on literature data (i.e. reports and scientific articles) and information received from stakeholders, while data on background processes (e.g. energy production, raw material production) were retrieved mainly from commercially available LCA databases (e.g. Ecoinvent and ELCD). All datasets were associated with (at least some degree of) variations, a point which was taken into consideration when performing the uncertainty analysis. Other important assumptions included: sorting efficiency at the sorting plants, the degree of oxidation of metals after thermal treatment, the quality of recyclables, the quality of substituted materials, emissions from the thermal processes and marginal energy production. The latter was the focus of the sensitivity

analysis, where alternative marginal electricity production processes were used to test the validity of the results in possible future energy scenarios.

Results

The handling of SR according to Scenario 1 (landfill) shows overall savings for Global Warming, Photochemical Oxidant Formation and Terrestrial Eutrophication, mostly because of the benefits associated with recovering and recycling plastic and aluminium fractions. For all non-toxic impact categories and within uncertainty, scenarios with increased energy recovery (i.e. S2-incineration and S4-cement kiln) from SR >4 mm show significantly better environmental performance compared with the baseline landfill-based scenario (i.e. S1), indicating that increased energy recovery is a desirable option for the non-toxic categories.

The results for the pyrolysis scenario (i.e. S3) are associated with significant uncertainty, mostly owing to the fact that the dataset for SR pyrolysis included rather broad ranges of data. This suggests that a clear conclusion on the environmental sustainability of pyrolysis as a treatment technology for SR is not possible with currently available data and understanding of the process. This means that further experimental investigations are needed before it can be concluded whether pyrolysis is a possible treatment technology for SR.

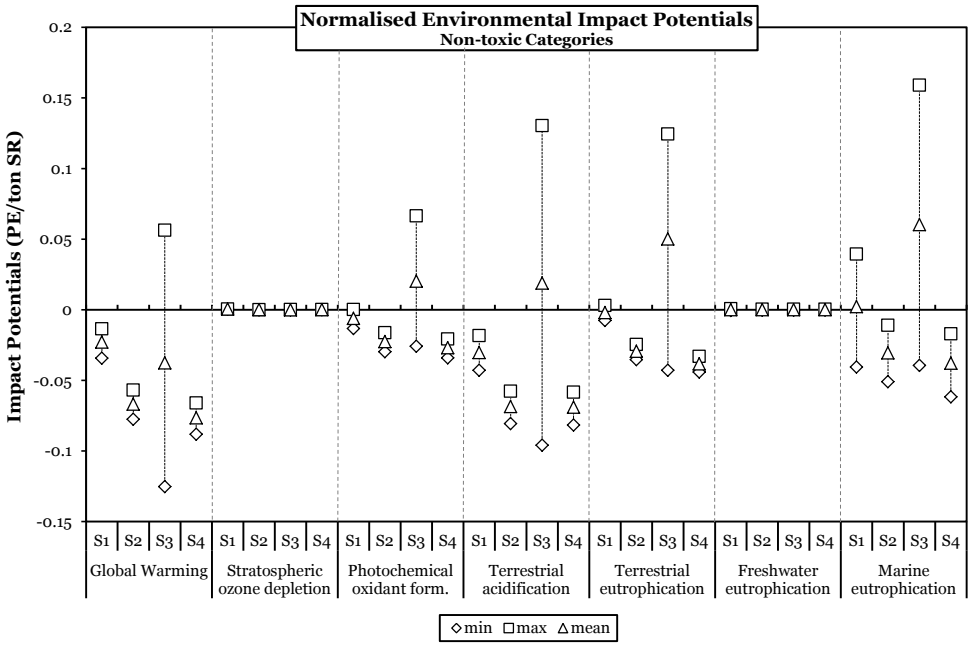


Figure A - Normalised potential non-toxic impact from treatment of 1 ton SR in the four assessed scenarios (PE = person equivalent), including min-max ranges.

Results for the potential toxic impacts show that the four analysed scenarios have comparable profiles for the impact categories Human Toxicity Carcinogenic, Particulate Matter and Ionizing Radiation. The three scenarios including the thermal treatment of >4 mm fractions all show increasing potential impacts on Human Toxicity non-Carcinogenic and Ecotoxicity compared with the baseline scenarios, owing to increases in emissions into the air, specifically As and Hg. In particular, the scenario based on co-combustion of >4mm in cement kiln (i.e. S4) presents potential impacts on Human Toxicity non-Carcinogenic and Ecotoxicity which are far larger than the other scenarios, owing particularly to emissions into the air of Cu, Hg and Zn. Compared to other technologies, these large impacts are connected with greater emissions, owing to less efficient flue gas cleaning at the cement plant and higher volatility of these metals due to higher temperature in the kiln. Such findings indicate the need for a pre-treatment step to produce an SR feedstock for the cement kiln with an RDF-like composition with a significantly lower content of metals.

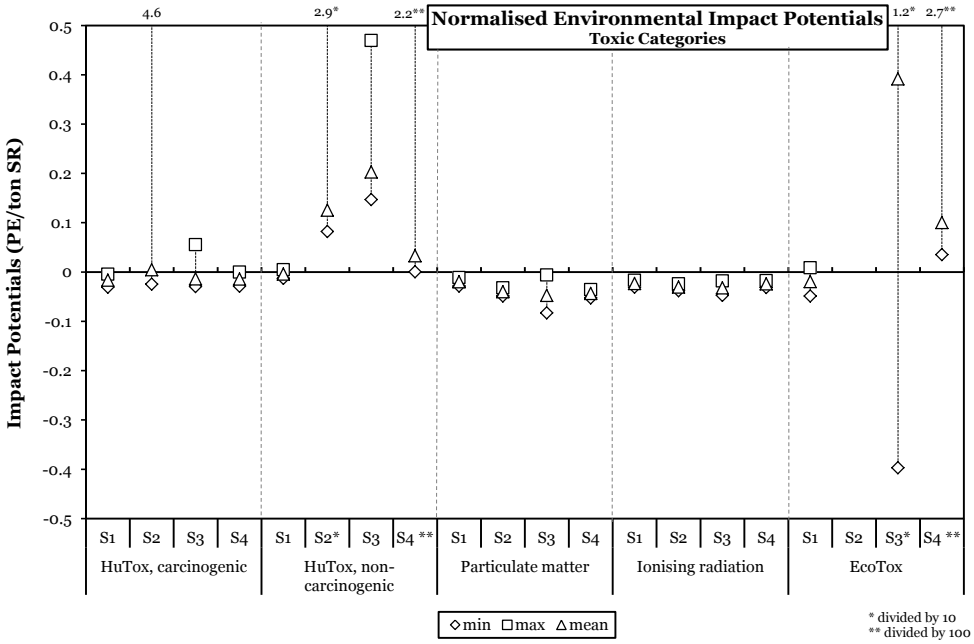


Figure B - Normalised potential toxic impact from treatment of 1 ton SR in the four assessed scenarios (PE = person equivalent), including min-max ranges.

When analysing the results by only looking at unitary input (i.e. 1 ton) of individual material fractions, normalised potential impacts on Global Warming indicate that recycling of both ferrous and non-ferrous metals provides the greatest savings, and it should thus be the first priority when analysing and designing a management scheme for SR. The next material is plastic, which can provide significant savings, especially if different resins are effectively sorted and the least possible amount of cross-contamination is achieved.

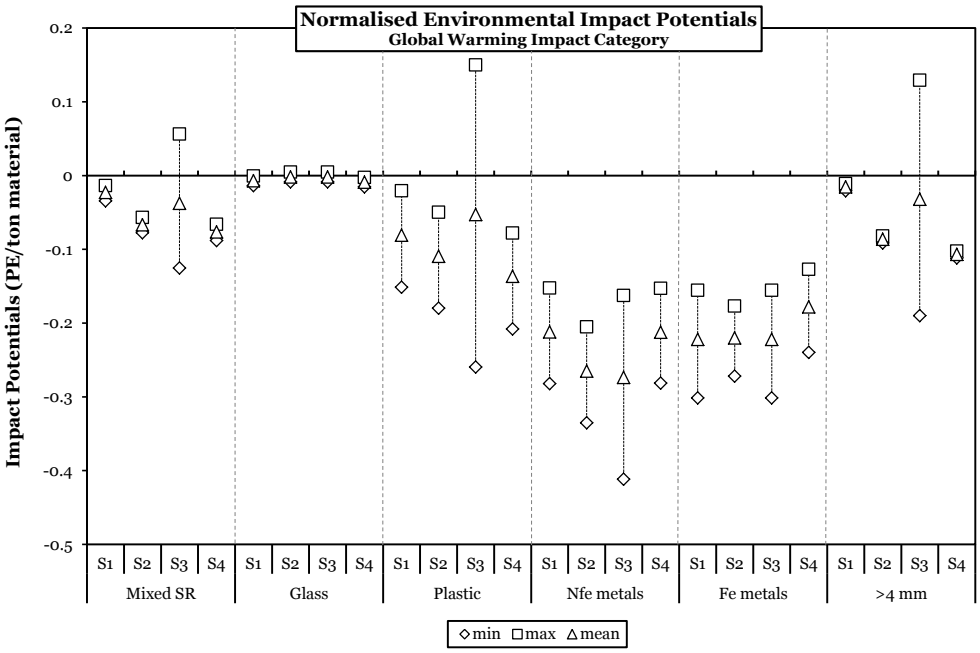


Figure C - Normalised potential impact on Global Warming in the four assessed scenarios, disaggregated according to individual material fractions and presented as per 1 ton of material (PE = person equivalent).

Sensitivity analysis

Sensitivity analysis was employed to test the influence of Danish marginal electricity production on the results, including possible future electricity production sources such as natural gas and wind, instead of coal. The results show that changes to marginal electricity production do not affect the ranking of scenarios in most impact categories, while absolute values for potential impacts were diminished, especially regarding savings from recycling processes and energy recovery.

The only exception was found in the Global Warming category, where scenario ranking was actually affected by marginal electricity technology. In fact, when offsetting natural gas, incineration provides similar savings to landfilling, while the scenario based on pyrolysis is close to not offering a net saving at all. When considering wind power as a marginal electricity production source, both scenarios based on incineration and pyrolysis showed worse performance than the scenario based on landfilling.

Conclusion

Based on data and current knowledge:

- Diverting SR >4 mm from landfill provides benefits from an environmental perspective and should thus be supported.
- Sorting and recycling metals, plastics and glass are beneficial to the environment and should thus be continued. Special focus should be given to increasing metal recovery, as this provides the greatest environmental benefits.
- Incineration seems currently the best option for the treatment of >4 mm residues.
- Pyrolysis seems to have worse energy efficiency but better downstream metal recovery (i.e. from the residues) than incineration. However, a clear conclusion could not be drawn, because results for pyrolysis were associated with significant uncertainty, owing to the lack of precise inventory data describing the process. Thus, any decision regarding the implementation of pyrolysis for treating >4 mm SR fractions should first be supported by pilot- to full-scale tests of the process, to deliver a better understanding thereof.
- The co-combustion of >4 mm fractions in a cement kiln could potentially provide significant savings for the Global Warming impact category. However, the current composition of the >4 mm fractions is not suitable, meaning that co-combustion in a cement kiln could only be implemented with a specially designed fraction. This option, in practice, would need a pre-treatment step to produce a feedstock with an RDF-like composition. This should especially include the additional screening and sorting of metals, to reduce the content of heavy metals in feedstock going into the kiln. Besides technical reasons, this would also result in a significant decrease in emissions into the air of heavy metals, which are largely responsible (especially Cu and Zn) for high potential impacts reported for some of the impact categories.
- Pre-treatment of the >4 mm SR should be considered, in order to reduce the content of heavy metals (especially Cu and Zn) in material fed into thermal processes.
- The results and recommendations are not significantly affected by the choice of the marginal technology, meaning they may still be valid within a different future energy system.

Sammenfatning og konklusion

Baggrund

Shredderaffald (Shredder residues - SR) er restfraktionen fra mekanisk sønderdeling af metalskrot som stammer fra forskellige kilder der indsamles på genbrugsstationer eller direkte hos metalforarbejdningsvirksomheder. Inden den mekaniske shredding, udsorteres først alle de dele der kan sendes direkte til genanvendelse og genbrug. Dansk SR er typisk en blanding af 20% shredded bilskrot (ASR) og 80% der stammer fra forskellige andre kilder (f.eks. hårde hvidevarer, demonterede både, industrielt storskrald og andet skrot). SR er i øjeblikket kategoriseret som farligt affald i Danmark og deponeres derfor, men lovgivningsmæssige og økonomiske forhold giver mulighed for at undersøge alternative løsninger til håndtering af SR.

Indledning

Nærværende rapport er udarbejdet af DTU Miljø og omfatter en livscyklusvurdering (LCA) af behandling af shredderaffald (SR) i Danmark. Vurderingen sammenligner de potentielle miljøpåvirkninger og forbug af abiotiske ressourcer i forbindelse med fire alternative scenarier, der alle inkluderer sortering af genanvendelige materialer og forvaltning af restmaterialer ved termisk behandling og / eller slutdeponering efter biologisk stabilisering.

Metode

Konsekvens LCA´en blev udført i henhold til ISO-standard 14040 og 14044, mens valget af LCIA metoder og påvirkningskategorier fulgte anbefalingerne i ILCD Håndbogen " Recommendations for Life Cycle Impact Assessment in the European context " (JRC, 2011). Systemudvidelse blev brugt til at vurdere multifunktionelle processer og til at godskrive undgået produktion ved genanvendelse, genvinding og nyttiggørelse. LCA´en blev udført ved hjælp af modellen EASETECH, som er udviklet på Danmarks Tekniske Universitet. Usikkerhedsberegningerne blev foretaget vha. Monte Carlo simulering med 10000 stokastiske variable til at beregne usikkerhedsfordelingen af resultaterne.

Scenarier

Fire scenarier til behandling af SR blev analyseret. Alle scenarier inkluderede sortering af genanvendelige materialer (dvs. glas, plast og metal) og deponering af fint restaffald (<4 mm), som er uegnet til termisk behandling på grund af dets høje indhold af tungmetaller. De enkelte scenarier inkluderede forskellig behandling af restfraktionen >4 mm, som var tilbage efter sortering af genanvendelige materialer. De 4 behandlingstyper for restfraktionen var: deponering i Scenarie 1 (S1), medforbrænding på et affaldsforbrændingsanlæg i Scenarie 2 (S2), pyrolyse i Scenarie 3 (S3), og medforbrænding i en cementovn i Scenarie 4 (S4).

Livscyklusopgørelser

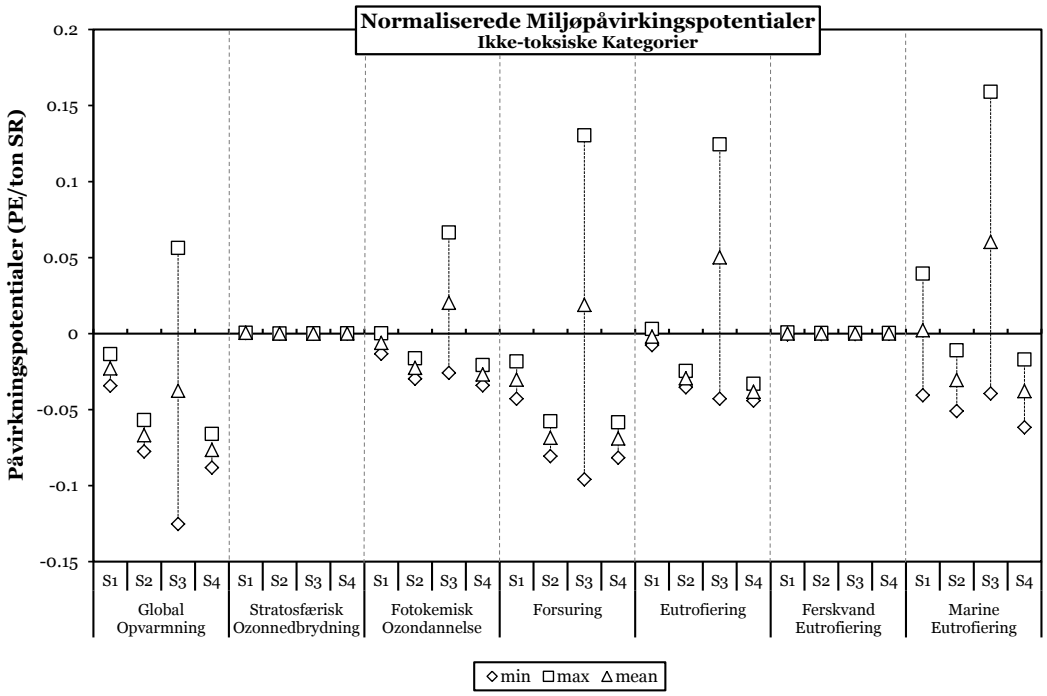
Datasæt vedrørende SR sammensætning og behandlingsteknologier blev hovedsageligt sammensat baseret på data fra litteraturen (dvs. rapporter og videnskabelige artikler), og oplysninger fra partnerskabet for projektet, mens data om baggrundsprocesser (f.eks. energiproduktion, råvareproduktion) hovedsageligt blev hentet fra kommercielt tilgængelige LCA-databaser (f.eks. Ecoinvent og ELCD). Alle datasæt havde en grad af variation, som blev taget i betragtning med de udførte usikkerhedsanalyser. Andre vigtige antagelser omfattede: sorteringseffektiviteten på sorteringsanlægget, graden af oxidation af metallerne efter termisk behandling, kvaliteten af materialer sendt til genanvendelse, kvaliteten af de substituerede materialer, emissioner fra termiske processer og den marginale energiproduktion. Sidstnævnte var fokus for en

følsomhedsanalyse, hvor alternative marginale el-produktioner blev brugt til at teste validiteten af resultaterne i mulige fremtidige energi-scenarier.

Resultater

Håndteringen af SR i Scenarie 1 (deponi) viser at der er besparelser i kategorierne Global Opvarmning, Fotokemisk Ozondannelse og Terrestrisk Eutrofiering, hvilket hovedsagelig skyldes besparelser forbundet med genindvinding og genbrug af plast- og aluminiums-fraktionerne. For alle ikke-toksiske påvirkningskategorier og inden for en margin af usikkerhed, findes det at scenarier med øget energiudnyttelse (dvs. S2-forbrænding og S4-cementovne) for SR >4 mm giver signifikant bedre miljøresultater sammenlignet med det deponi-baserede basisscenarie (dvs. S1). Dette indikerer at øget energigenvinding er en oplagt mulighed for de ikke-toksiske kategorier.

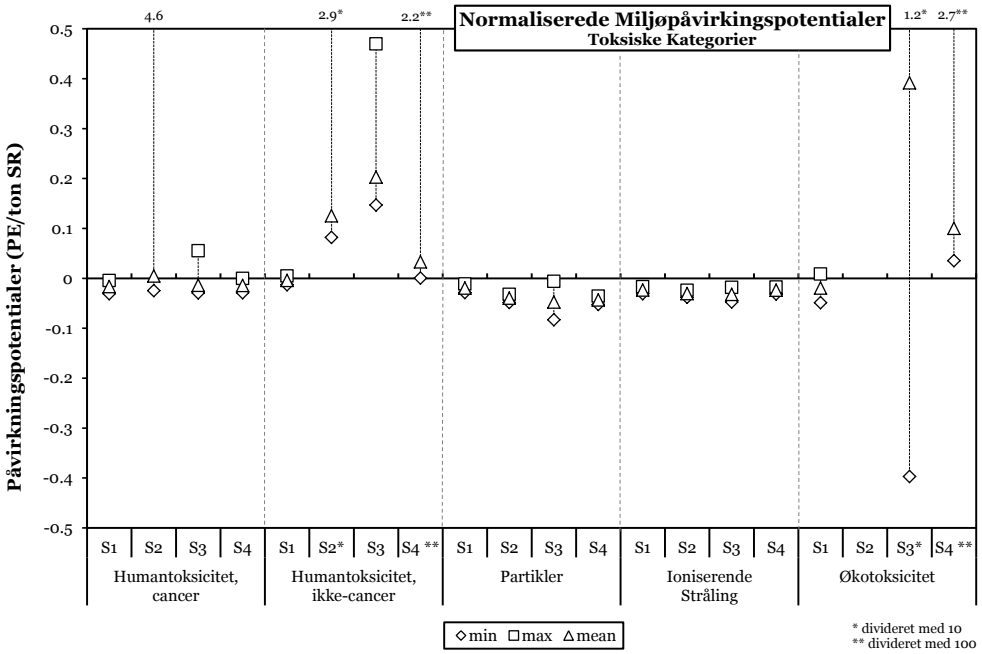
Resultaterne for pyrolyse scenariet (dvs. S3) er forbundet med betydelig usikkerhed, mest på grund af det faktum, at datasættet for SR pyrolyse inkluderer temmelig store intervaller af data. Dette peger på, at en klar konklusion ang. den miljømæssige bæredygtighed af pyrolyse som en behandlingsteknologi til SR ikke er mulig, med de tilgængelige data og viden omkring processen. Dette betyder, at der er behov for yderligere eksperimentelle undersøgelser, før det kan konkluderes om pyrolyse kan være en mulig behandlingsteknologi for SR.



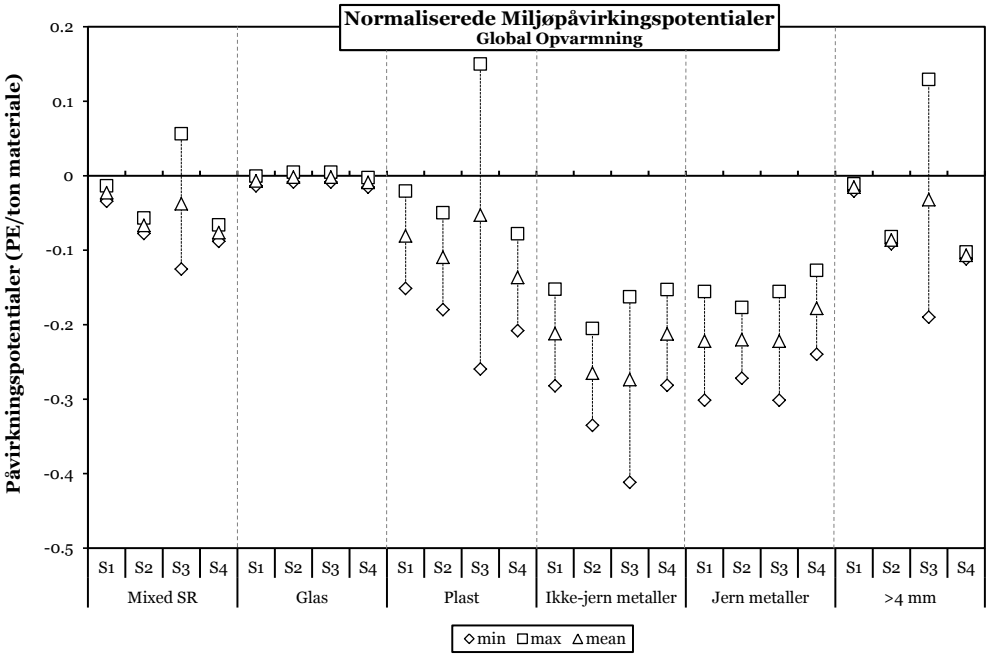
Figur A - Normaliserede potentielle ikke-toksiske miljøpåvirkninger fra behandling af 1 ton SR i de fire vurderede scenarier (PE = person ækvivalent), min-max intervaller er inkluderet i figuren.

Resultaterne for de potentielle toksiske påvirkninger viser, at de fire analyserede scenarier har sammenlignelige profiler for påvirkningskategorierne Humantoksicitet (kræftfremkaldende), Partikler og Ioniserende Stråling. De tre scenarier, herunder termisk behandling af >4 mm fraktionen viser øgede potentielle påvirkninger for kategorierne Humantoksicitet (ikke-kræftfremkaldende) og Økotoksicitet sammenlignet med de grundlæggende scenarier, som følge af stigninger i emissionerne til luften, navnlig As og Hg. Scenariet baseret på medforbrænding af >4 mm SR i cementovn (dvs. S4) bidrager med potentielle påvirkninger til Humantoksicitet (ikke-kræftfremkaldende) og Økotoksicitet, der er langt større end de øvrige scenarier især på grund emissioner til luften af Cu, Hg og Zn. De øgede påvirkninger skyldes mindre effektiv røggasrensning på cementfabrikken. Dette forværres yderligere af at disse stoffer er mere flygtige i cement processen grundet de højere temperaturer, hvilket gør at der er væsentlige højere udledninger.

Grundet den høje påvirkning fra metallerne, er der behov for forbehandling af SR for at producere en råvare til cementovne med en RDF-lignende sammensætning, med et lavere metalindhold.



Figur B - Normaliserede potentielle toksiske miljøpåvirkninger fra behandling af 1 ton SR i de fire vurderede scenarier (PE = person ækvivalent), min-max intervaller er inkluderet i figuren.



Figur C - Normaliserede potentielle påvirkninger for Global Opvarmning for de fire opstillede scenarier, opdelt efter de enkelte materialer og præsenteret per 1 ton materiale (PE = person ækvivalent).

Hvis man i stedet ser på en enkelt fraktion (dvs. 1 ton af de enkelte materialefraktioner), viser de normaliserede potentielle påvirkninger i forhold til Global Opvarmning, at genanvendelse af magnetiske og ikke-magnetiske metaller giver de største besparelser, og det bør derfor være første prioritet at udsortere disse, ved planlægning af håndteringen af SR. Efter metallerne er plast et materiale som kan give betydelige besparelser, især hvis forskellige plastiktyper effektivt udsorteres og den mindst mulige mængde af krydskontaminering opnås.

Følsomhedsanalyse

Følsomhedsanalyser blev brugt til at teste indflydelsen af den danske marginale elproduktion på resultaterne, herunder eventuelle fremtidige energikilder til elproduktion, såsom naturgas og vind, i stedet for kul. Resultaterne viser, at ændringen af den marginale elproduktion ikke påvirker rangordenen af scenarier i de fleste påvirkningskategorier, mens absolutte værdier for potentielle påvirkninger blev mindsket, især med hensyn til besparelserne fra genbrugsprocesser og energiudnyttelse.

Den eneste undtagelse blev fundet til at være påvirkningerne i kategorien Global Opvarmning, hvor rangordenen af scenarier blev påvirket af den marginale el-teknologi. Når naturgas erstattes giver forbrænding tilsvarende miljømæssig besparelse i forhold til deponering, mens scenariet baseret på pyrolyse giver meget lav nettobesparelse. Hvis vindkraft er den marginale elektricitetskilde giver begge scenarier med forbrænding og pyrolyse dårligere resultater end scenariet med deponering.

Konklusion

Baseret på data og viden fra dette studie:

- Ændring af den nuværende behandling af SR >4 mm således at mere genanvendes og mindre deponeres, giver fordele fra et miljømæssigt synspunkt og bør derfor prioriteres.
- Sortering og genanvendelse af metaller, plast og glas er til gavn for miljøet og bør derfor fortsættes. Der bør være særligt fokus på at genanvende metaller, da dette giver de største miljømæssige fordele.
- Forbrænding er miljømæssigt den bedste mulighed for behandling af SR > 4 mm.
- Pyrolyse har dårligere energieffektivitet, men bedre nedstrøms genvinding af metaller (dvs. fra restprodukter) end forbrænding. Dog kunne en klar konklusion ikke drages, fordi resultaterne for pyrolyse var forbundet med væsentlig usikkerhed på grund af manglen på præcise data, der beskriver processen. Således bør enhver beslutning vedrørende gennemførelsen af pyrolyse til behandling af SR fraktion >4 mm blive understøttet af pilot-til-fuldskala test af processen, for at levere et bedre datagrundlag for processen.
- Medforbrænding af >4 mm fraktionen i cementovne kan potentielt give en betydelig besparelse for påvirkningskategorien Global Opvarmning. Den nuværende sammensætning af fraktionen >4mm er dog ikke egnet, da med-forbrænding i cementovne kun kan gennemføres med en specielt designet fraktion. Denne mulighed betyder inkludering af forbehandling for at producere et råmateriale med en RDF-lignende sammensætning. Dette vil især omfatte en yderligere screening og sortering af metaller, for at reducere indholdet af tungmetaller i råmaterialet som sendes til cementovne. Udover tekniske grunde, vil det også resultere i en betydelig reduktion af emissioner til luft af tungmetaller især Cu og Zn som i høj grad er grund til høje potentielle påvirkninger rapporteret for nogle af påvirkningskategorierne.
- Forbehandling af >4 mm SR bør overvejes for at reducere indholdet af tungmetaller (især Cu og Zn) i inputtet til termiske processer.
- Resultaterne og anbefalingerne påvirkes ikke væsentligt af valget af den marginale energiteknologi, hvilket betyder at de stadig kan være gældende for et alternativt fremtidigt energisystem.

List of abbreviations

AC	Acidification
AE	Aquatic Ecotoxicity
ARD	Abiotic Resource Depletion
ASR	Automotive shredder residues
EU	Eutrophication
EQ	Ecosystem Quality
FEW	Freshwater Ecotoxicity
GW	Global Warming
ILCD	International Reference Life Cycle Data System
HH	Human Health
HT	Human Toxicity
LCA	Life cycle assessment
LCI	Life cycle inventory
LCIA	Life cycle impact assessment
LU	Land Use
MSW	Municipal solid waste
OD	Ozone Depletion
POF	Photochemical Ozone Formation
RES	Resource Depletion
SR	Shredder residues
TC	Transfer coefficients
TE	Terrestrial Ecotoxicity

1. Introduction

1.1 Background

Shredder residue (SR) is the "residual fraction from mechanical shredding of metal containing scrap originating from different sources and processed at recycling stations or metal recovery businesses" (Hyks et al., 2014). Danish SR is typically a mixture of 20% automotive shredder residue (ASR) and 80% other residues originating from different sources (e.g. white goods, dismantled boats, industrial bulky waste, metal scrap). SR is a heterogeneous mixture of different materials, such as rubbers, plastics, glass, wood, dusts, fibres, metals, minerals and other residues that are difficult to separate. SR accounts for approximately 20% of the material input into sorting and recycling facilities, and because of the high content of some metals and different organic contaminants, it is classified as hazardous waste in Denmark (Hyks et al., 2014).

While the majority of SR in Denmark (and also in Europe) is currently disposed of in hazardous waste landfills, alternative management solutions are being explored and implemented. This is a legislative- and market-driven process, owing to amongst other reasons:

- Landfilling of hazardous waste in Denmark is subject to increasing taxation rates: the current tax of 160 DKK per tonne of landfilled SR will be increased to 475 DKK from January 1, 2015 (Skatteministeriet, 2011).
- EU Council Directive 2000/53/EC on ELV mandates the increasing reuse and recycling of SR: the current target of 85% will increase to 95% by 2015, where mechanical processes must encompass at least 85% of the SR and thermal processes may be used for up to 10% of the total amount (European Council, 2000). A target of >85% can be achieved by reusing components or recycling raw materials into new products.
- Generally, increasing raw material and energy prices make recovery and recycling economically more practicable and interesting.

1.2 Objectives of the project

The purpose of this project is to perform a life-cycle assessment (LCA) of the management of Danish SR. The project deals with newly produced SR. The starting point for the management system is sorting and recycling, followed by energy recovery and finally the disposal of the generated residues. As the majority of SR is currently sent to landfill, where it is then stored, the goal of this LCA is to assess the possible treatment options and the environmental consequences from treating the residue and recovering materials.

Considering the large variations in data, LCA results are calculated and presented by using ranges for input data values, thereby providing recommendations that include uncertainty. The assessed scenarios are developed by taking into consideration their technological feasibility within legislative constraints regarding the use and disposal of secondary materials as a result of the treatment. The results of the present LCA should provide a basis for deciding on the future management of SR, in order to ensure increased resource exploitation thereof.

1.3 Existing LCA studies

Table 1 provides an overview of existing LCA studies addressing the management of SR and ASR. The reviewed studies mainly cover Europe and focus almost entirely on ASR. Despite their limited

number, the available studies indicate that increasing the recovery of materials and energy provides significant environmental benefits compared to landfilling SR. However, the preferred technological solution varies between studies, depending on factors such as residue composition, local energy mix and technological capability.

- Weaknesses and limitations in current existing studies (Table 1) can be summarised as follows:
- No studies provide life cycle inventory (LCI) modelling based fully on the material fractions and physicochemical compositions of SR.
- No studies provide a comprehensive and systematic sensitivity/uncertainty analysis testing the robustness of results while also highlighting the key parameters influencing the results.

Table 1 - Overview of existing LCA studies on the management of SR and ASR.

Geographical area	Type	Composition	Scenarios included	Impact categories	Uncertainty analysis	Results	Source
US	ASR	Material fraction + partial chemical	- Landfill - Thermal recycling - Cement kiln - Material recycling	AC,EU, FEW,GW,HT,POF, TE	None	Scenarios 2,3,4 better than scenario 1	/A/
Italy	ASR	Material fraction	- Landfill - Non-Ferrous recovery - Co-incineration - Plastic recovery + co-incineration - Chemical recycling (methanol)	HH,EQ, RES	Sensitivity + propagation	Scenario 4 and 5 are best	/B/
Belgium	ASR	Chemical	- Landfill - Recycling + landfill - Incineration - Recycling + incineration	LU GW HT	None	Recycling + energy recovery provides largest benefits	/C/
DK	SR	Material fraction	- Landfill - Sieving + pyrolysis - Pyrolysis	GW,OD, AC,TE,AE, Waste	Scenario analysis on marginal energy	Scenario 2 best	/D/
Italy	ASR	Material fraction	- Landfill - Incineration - Mechanical recycling - Chemical recycling (methanol)	HH,EQ, RES	Monte Carlo	Scenario 3-4 best	/E/
Portugal	ASR	Material fraction	- Landfill - Incineration - Additional dismantling + SRF + cement kiln	ARD,GW,POF,AC,EU	Parameter sensitivity	Scenario 3 best	/F/

Midpoint → AC: Acidification; AE: Aquatic Ecotoxicity; ARD: Abiotic Resource Depletion; EU: Eutrophication; FEW: Freshwater Ecotoxicity; GW: Global Warming; HT: Human Toxicity; LU: Land Use; OD: Ozone Depletion; POF: Photochemical Ozone Formation; TE: Terrestrial Ecotoxicity.
Endpoint → HH: Human Health; EQ: Ecosystem Quality; RES: Resource Depletion

/A/: Boughton et al. (2006); /B/: Ciacci et al. (2010); /C/: Vermeulen et al. (2010); /D/: Høstgaard et al. 2012; /E/: Passarini et al. (2012); /F/: Fonseca et al. (2013).

2. Scope and design of the LCA

2.1 General principles

The life cycle assessment (LCA) in this project is performed as a 'consequential LCA', whereby environmental impacts are calculated in relation to the management of SR. In accordance with the consequential approach, the assessment makes use of system expansion to credit for any material production avoided and for energy in the background system.

The LCA is conducted under the principle of 'best practice', regarding the choice of LCIA methods and environmental impact categories and the assessment of data quality. The LCA is performed according to ISO standards 14040 and 14044. The selection of LCIA methods and environmental impact categories follows recommendations stipulated in the ILCD Handbook 'Recommendations for Life Cycle Impact Assessment in the European Context' (JRC, 2011).

2.2 The functional unit

The functional unit defines the service provided by the system assessed by the LCA study, thus ensuring that all the assessed waste management scenarios are comparable, meaning that they provide an equivalent service. The functional unit of the present LCA is defined as:

Treatment of 1 ton of newly produced shredder residues (SR), including treatment, recycling, incineration/pyrolysis and final disposal of any residues from the treatment process. Composition of SR under assessment is specified in Chapter 3.2, while inventory data for individual treatment processes are presented in Chapter 4.1.

The time horizon for the assessment is assumed to be 100 years, while the geographical scope of the SR treatment processes is assumed as being in Denmark and the geographical origins of individual background processes are specified in Appendix 1.

2.3 System boundaries

Figure 1 presents the system boundary of the LCA and the overall material flow. SR enters the system boundaries burden-free, meaning that upstream processes and impacts related to the manufacturing of products becoming SR are not included. In all scenarios, SR is sorted to recover recyclable materials, while the remaining material is sent for energy recovery (incineration, pyrolysis or co-combustion in a cement kiln) and/or deposition after biological stabilisation. Substitutions for virgin materials and energy are shown in the dashed boxes in Figure 1.

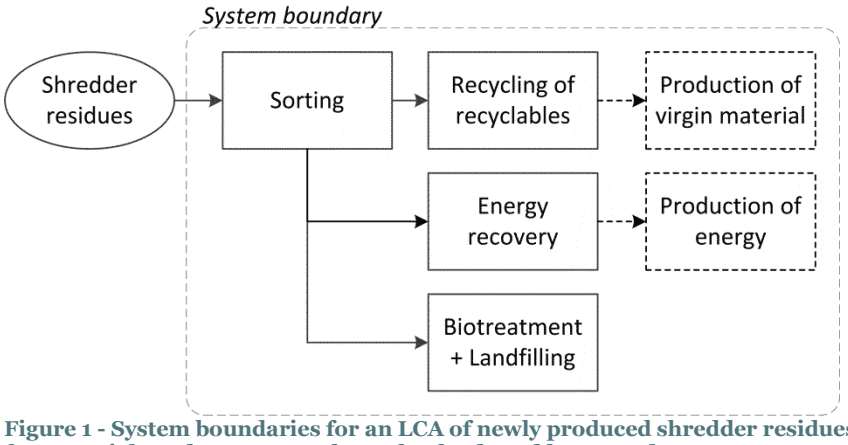


Figure 1 - System boundaries for an LCA of newly produced shredder residues. The substitutions for materials and energy are shown by the dotted boxes and arrows.

2.4 Allocation/system expansion

According to recommendations for state-of-the-art LCAs (JRC, 2011), system boundary expansion was applied within the current project, in order to ensure a consistent approach and a holistic perspective to support decision making. In practice, this has two consequences: 1) cascading effects (i.e. lost opportunities) on other systems, owing to the decision under assessment, are included within the system boundaries, and 2) substitution is used instead of allocation whenever a multi-output process is modelled, meaning that the waste system was credited for avoiding emissions that would otherwise have occurred in the production of the substituted products. The latter is also needed to ensure equivalency amongst the assessed scenarios.

2.5 Environmental impact categories, LCIA methods and interpretation of the results

The present LCA makes use of environmental impact categories and LCIA methods described in the ILCD Handbook (JRC, 2011). In addition, the impact category Depletion of Abiotic Resources (fossil and elements) from the CML method is included for both fossil and elemental resources independently, whereas in the guidelines an aggregated impact is used. Other included categories are Human Toxicity and Eco Toxicity in the form of USEtox, which was also recommended in the ILCD Handbook (JRC, 2011). Particulate substances that affect breathing, however, are not included in the USEtox category for Human Toxicity, so they are therefore modelled according to the UPFM model (Humbert , 2009). An overview of the selected environmental impact categories is shown in Table 2.

It should be noted that USEtox is subject to some uncertainty, which should be considered when interpreting the results (an update to USEtox version 1.1 is in preparation, especially with an improved modelling of the speciation of the metals).

In this LCA, the potential environmental impacts will be converted for each of the impact categories into a single entity in the form of a person equivalent (PE), with the actual load divided by the average annual load produced by one person – referred to as 'normalisation'. Table 2 also shows the applied normalisation conversion equivalents for the impact categories used herein, and these normalisation references are based on Blok et al. (2013).

Table 2 – Environmental impact categories and normalisation references used in the present project.

Impact category	Method	Abbreviation	Normalisation reference	Unit
Global Warming	IPCC 2007	GWP100	8100	kg CO2-eq./PE/year
Stratospheric Ozone Depletion	WMO 1999	ODP	$4.14*10^{-2}$	kg CFC11-eq./PE/year
Human Toxicity, Cancer Effect[1]	USEtox	HT-C	$5.42*10^{-5}$	CTUh /PE/year
Human Toxicity, non-Cancer Effect[1]	USEtox	HT-NC	$1.10*10^{-3}$	CTUh/PE/year
Particulate Matter	Humbert	PM	2.76	kg PM 2.5/PE/year
Ionizing Radiation, Human Health	Dreicer		$1.33*10^{3}$	kBq U-235 air-eq/person
Photochemical Ozone Formation	ReCiPe midpoint		56.7	kg-NMVOCeq/person
Terrestrial Acidification	Accumulated Exceedance	AP	49.6	AE/PE/year
Terrestrial Eutrophication	Accumulated Exceedance	TEP	115	AE/PE/year
Freshwater Eutrophication	ReCiPe midpoint	FEP	0.62	kg P-eq./PE/year
Marine Eutrophication	ReCiPe midpoint	FEP	9.38	kg N-eq./PE/year
Freshwater Ecotoxicity[2]	USEtox	ET	$6.465*10^{-5}$	CTUe/PE/year
Depletion of Abiotic Resources-Fossil	CML	ADP-F	$6.24*10^{-4}$	MJ/PE/year
Depletion of Abiotic Resources-Elements	CML	ADP-E	$3.43*10^{-2}$	kg Sb-eq./PE/year

[1] CTU_h comparative toxic unit for humans.
[2] CTU_e - comparative toxic unit for ecosystem.

2.6 The EASETECH Life Cycle Assessment Tool

Life cycle modelling was facilitated with the EASETECH waste LCA model (Clavreul et al., 2014), developed by the Technical University of Denmark. EASETECH allows for estimating mass flows, resource consumption and emissions from the waste system under assessment, based on a detailed physicochemical composition of material fractions in the waste and a number of user-defined process parameters. EASETECH includes specific modules for modelling source separation, the collection and transportation of waste, material recycling facilities, incineration, composting, biogas, combined biogas and composting plants, landfills, the use of stabilised organic waste in agriculture, the recycling of materials and energy and materials production.

The EASETECH model contains ready-to-use data for a number of selected processes and technologies, while the inventory modelling of specific plants can be performed and its results stored in the model. Furthermore, scenarios with plural routing options can be established for any kind of waste system, starting with waste generation and ending with final disposal in a long-term repository, the industrial recycling of materials, the spreading on agricultural land of bio-treated organic waste, use in power plants or material utilisation. Owing to the market effects induced by material recycling, energy recovery or material utilisation, the waste system can be credited for resource and environmental savings achieved through the corresponding avoided production of virgin materials. EASETECH also contains databases for a number of key processes, such as transport, electricity and heat production, and inventory data can be imported from commercial databases.

2.6.1 Uncertainty calculations

Uncertainty calculations are carried out via Monte Carlo simulation. In EASETECH, this is done by specifying distributions for each parameter for which the uncertainty is to be assessed. The model accepts a mixture of normal, lognormal, triangular and uniform distributions. After having specified all the parameter distributions for which the Monte Carlo analysis is to be carried out, the user needs to specify how many random variables need to be used within each distribution. In this report we used 10,000 variables.

2.7 Data needs

2.7.1 Material composition

Information regarding the material fraction and physicochemical composition of SR was retrieved from Hansen et al. (2011a) and Høstgaard et al. (2012). As shown in Table 16 and Table 17 of Appendix 2, a range of other studies providing compositional data were available, but they could not be utilised in the present context because they dealt with either ASR (which is only 20% of Danish SR) or SR excavated from landfills (e.g. Hansen et al., 2012; Ahmed et al., 2014). A detailed overview of the physicochemical composition of SR used in LCA modelling is provided in Chapter 3.2 and Appendix 2.

2.7.2 Treatment of shredder residues

For all SR treatment processes, specific inventory datasets were established, as presented in detail in Chapter 4.1. For the remaining background processes involved in the LCA, datasets from recognised LCA databases were used. A comprehensive list of the datasets used in the present study is provided in Chapter 4.3.

2.8 Non-quantitative assessments

In the present study, potential impact weighting is not performed, as it is not required by the ISO standard. A qualitative evaluation of the uncertainty related to characterisation factors and methods is mentioned in Chapter 5, with particular emphasis placed on separately discussing non-toxic and toxic impact categories during the interpretation of results.

2.9 Limitations

The present LCA study is performed in accordance with ISO standards 14040 and 14044, and it is generally meant to be used as a basis for decision-making. In this context, it should be emphasised that the LCA should not be used as the only tool for decision support. The results of the present project are valid as long as the socio-demographic and technological conditions of the SR management system (described later) persist.

2.10 Requirements for data quality

To evaluate the quality of external processes included in the LCA, the methodology developed by Weidema & Wesnæs (1996) was used. This method uses quality indicator values for five indicator categories to assess the quality of data. The indicator categories include credibility, completeness, temporal correlation, geographical correlation and technological correlation. For individual categories, indicator values are assigned by comparing the quality of process data with data quality goals set according to the goal and the scope of the project. A value is assigned on a scale of 1 to 5, where 1 indicates full documentation and compliance, and 5 indicates a lack of data/compliance with the required process. The average of the indicator categories is then calculated. Whenever possible, this project makes use only of processes which have an average indicator value less than 3. A definition of the indicators, and an explanation of the individual indicator categories, is available in the documentation prepared for the Ecoinvent database (Frischknecht et al., 2007), initially based on Weidema & Wesnæs (1996).

2.11 Process regarding critical review

The report was reviewed externally by LCA experts from the Danish Technological Institute. The review report is available in Appendix 9. All comments from the reviewers have been implemented in this version of the report.

2.12 Report format

The format of the report follows the recommendations provided by the ISO 14040 and 14044 standards.

2.13 Timeframe

Data on the composition of newly produced SR refer to the years 2011-2012. The composition of SR is expected to change over the years; for example, the use of composite plastic materials is increasing in the automotive industry (Koronis et al., 2013). The validity of SR composition data will thus have to be checked when developing future scenarios. Inventory data for processing systems refer to current average technologies; hence, the results of the current study are only valid as long as there are no substantial changes in the design of technologies and background systems. The latter applies especially to marginal technology for energy production. As Denmark is in a transition phase and heading towards a fossil-free energy system, the relevance of energy recovery from SR may change significantly in the future (e.g. in ten years).

3. Scenarios and composition of shredder residues

3.1 Scenarios

The scenarios included herein are illustrated in Figure 2 to Figure 5. The main difference between the individual scenarios is the technology (i.e. landfilling, incineration, pyrolysis and co-combustion in a cement kiln) employed for treating >4 mm residue remaining after sorting recyclable materials. All scenarios include the sorting of recyclables and the landfilling of fine residues <4 mm, which are not suitable for thermal treatment because of their high content of heavy metals.

Scenario 1 (S1) represents the baseline current situation for the management of SR (Figure 2). In this scenario, recyclables (i.e. glass, plastics and metals) are sorted and recovered from SR, while the remaining residues are landfilled. Recyclables are sent to recycling facilities to produce secondary glass, plastic and metal materials.

To illustrate the effect of treating the remaining residues biologically before landfilling, this technology step has been included in all scenarios, despite the fact that this treatment method is currently not applied in Denmark. More details are found in Møller et al. (2014), where it is shown that biological treatment has almost no effect on the results of the LCA.

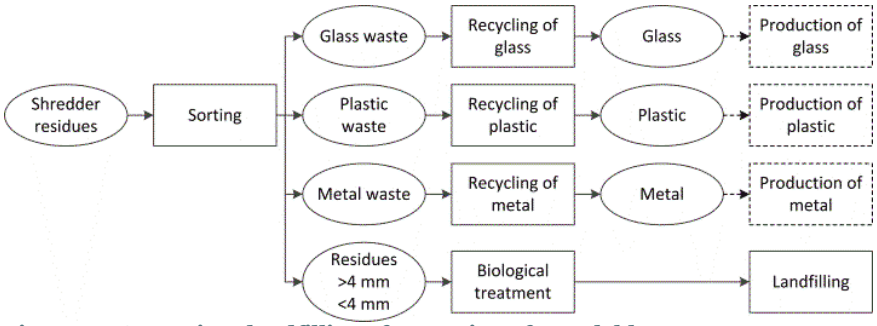


Figure 2 – Scenario 1: landfilling after sorting of recyclables.

Scenario 2 (S2) includes the recovery of recyclables (i.e. glass, plastics and metals) from SR (Figure 3). The remaining residues are then screened to separate particle fractions >4 mm, which are routed to incineration, and a fraction <4 mm, which is sent for biological treatment and then to landfill. Recyclables are sent to recycling facilities for the production of secondary glass, plastic and metal materials. Bottom ashes from the incineration process are treated to enable the further recovery of metals, and then they are used for road construction as sub-base materials. Fly ashes from the air-pollution-control (APC) system of the incineration plant are landfilled.

Scenario 3 (S3) includes the recovery of recyclables (i.e. glass, plastics and metals) from SR (Figure 4). The remaining residues are then screened to separate a particle fraction >4 mm, which is routed to pyrolysis, and a fraction <4 mm, sent for biological treatment and then to landfill. Metals are recovered from pyrolysis char, which is then used together with pyrolysis oil and gas as a fuel for energy production in incineration plants. Recyclables are sent to recycling facilities for the

production of secondary glass, plastic and metal materials. Bottom ashes from combustion of pyrolysis products (Char etc.) presumably are used for road construction as sub-base materials. Fly ashes from the APC system are landfilled.

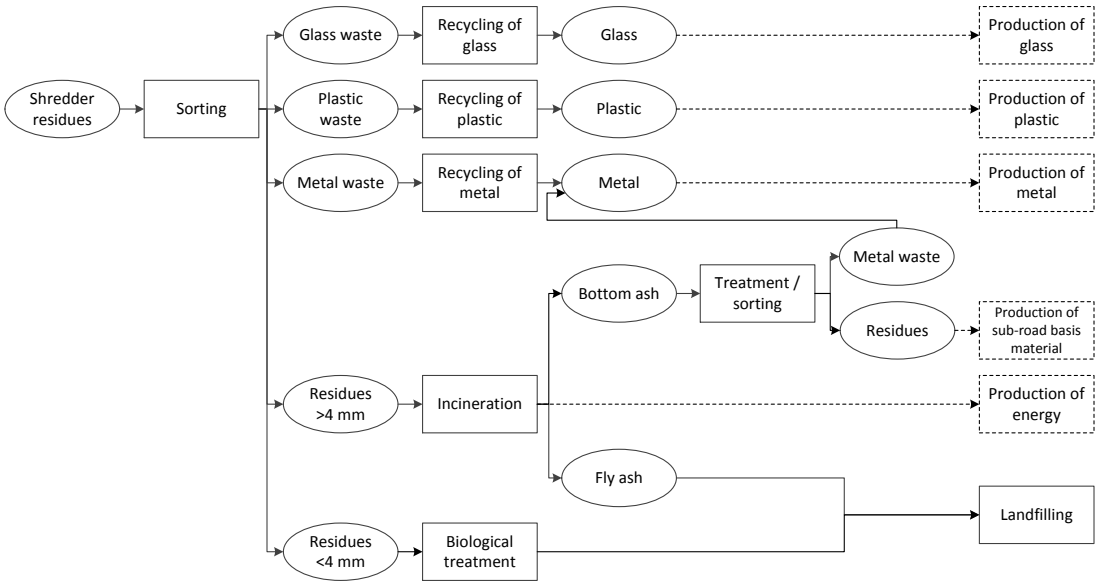


Figure 3 – Scenario 2: incineration of >4 mm residues after sorting of recyclables.

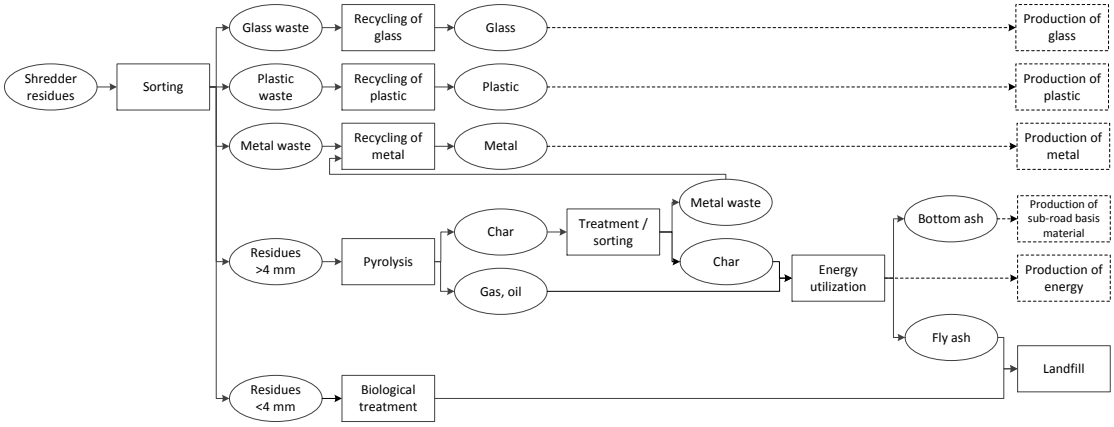


Figure 4 – Scenario 3: pyrolysis of >4 mm residues after sorting of recyclables.

Scenario 4 (S4) includes the recovery of recyclables (i.e. glass, plastics and metals) from SR (Figure 5). The remaining residues are then screened to separate a particle fraction >4 mm, which is used as a co-fuel in cement kilns, and a fraction <4 mm, sent for biological treatment and then to landfill. While cement kilns can generally run using different fuels (i.e. coal, petcoke, natural gas, oil), almost all European and Danish plants make use of coal (Daugaard, 2014). When coal is substituted with an alternative fuel such as waste, 8% additional energy is needed in the process. In fact, as waste has a high content of chlorine compared with coal, some of the off gas needs to be drained from the kiln. This results in a loss of energy in the system, which needs to be compensated for through the additional input of fuel (Daugaard, 2014). For LCA modelling, it was thus assumed that 1 MJ of SR substitutes 0.926 MJ of coal (or 1.08 MJ of SR substitutes 1 MJ of coal). Inert materials in SR end up in the final cement product, while fly ashes from the APC system are landfilled.

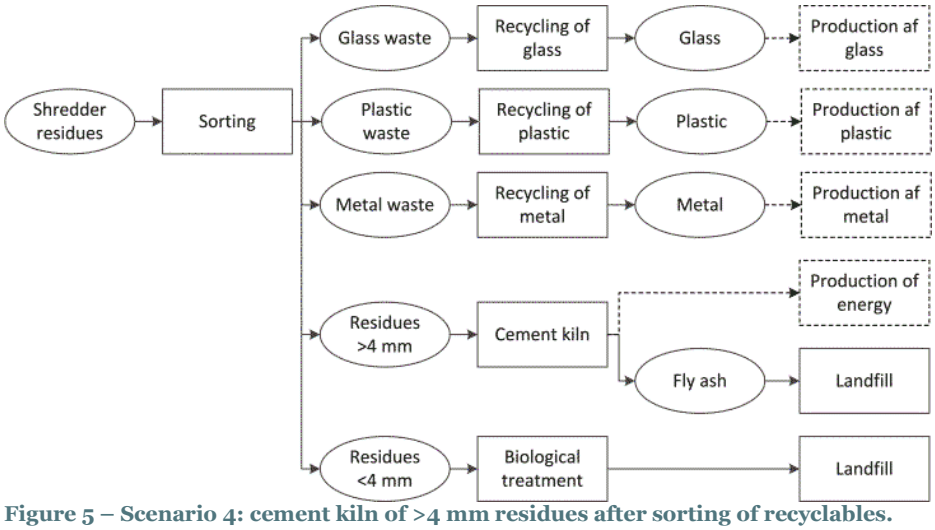


Figure 5 – Scenario 4: cement kiln of >4 mm residues after sorting of recyclables.

The report does not include an analysis of possible transport distances, as there is no knowledge of where the material will be treated, while, at least for the fractions of recyclable materials, transportation would be the same in all scenarios. In addition, it is assumed that emissions from transport will only play a very minor role with regards to environmental impacts (Merrild et al., 2012).

3.2 Composition of shredder residues

Waste is generally a heterogenic material, but some waste – such as SR – varies significantly over days and even hours. The composition of SR depends on material received at the shredder companies and the sorting performed, which in turn depends on the market price of waste materials and recyclables. Intervals for the content of individual materials were thus used in this assessment, in order to cover the heterogeneity of SR.

The material fraction composition used in the LCI modelling is presented in Table 3, based on data from Hansen et al. (2011a) and Høstgaard et al. (2012). These data are in line with recent results from two sampling/sorting campaigns conducted by Stena Recycling in April 2014 (undisclosed because of confidentiality reasons). Details about dataset creation and the uncertainty assessment are presented in Appendix 2.

The physicochemical composition of <4mm and >4mm fractions of SR used in the modelling is presented in Table 4. These datasets are based on data from Hansen et al. (2011a) and Høstgaard et al. (2012) and are presented as lognormal distributions (additional details in Appendix 2).

Table 3 – Material fraction composition used for LCI modelling SR.

Fraction	Average (%)	St. Dev.
Ferrous metal	0.93	0.53
Non-ferrous metal	3.22	0.99
Tin can	0.67	0.21
Plastic	11.88	2.86
Rubber	6.89	2.10
Glass	0.03	0.01
>4mm	30.83	3.72
<4mm	45.54	3.15
Total	100	

Table 4 – Physicochemical composition of <4mm and >4mm fractions of SR used in the LCI modelling (based on Hansen et al. (2011a) and Høstgaard et al. (2012)). All data are provided as lognormal distributions.

Parameter	Unit	Fine <4mm		> 4mm	
		median	deviation	median	deviation
TS*	% ww	89.4	3.25	89.4	3.25
Ash	% TS	80.9	21.6	40.4	21.4
LHV	MJ/kgTS	0.33	4.12	17.1	6.33
C	% TS	5.09	9.26	35.4	6.89
H	% TS	2.60	1.90	4.96	1.61
N	% TS	0.49	0.16	0.83	0.23
Si	mg/kg TS	103000	66300	50500	38100
Al	mg/kg TS	28700	17300	16600	15500
Ca	mg/kg TS	50000	41500	33100	14900
Fe	mg/kg TS	192000	88200	73700	44500
K	mg/kg TS	5340	4610	3000	1040
Mg	mg/kg TS	9210	5670	7110	1540
Mn	mg/kg TS	2070	1110	1040	506
Na	mg/kg TS	11600	6290	6860	3030
P	mg/kg TS	1440	2780	1090	767
Ti	mg/kg TS	5470	4160	4900	4220
As	mg/kg TS	88.4	286	49.2	107
Ba	mg/kg TS	3170	1560	2540	2250
Be	mg/kg TS	0.644	0.187	0.615	
Cd	mg/kg TS	25.2	41.0	9.73	
Co	mg/kg TS	41.1	38.8	35.8	
Cr	mg/kg TS	979	1950	391	272
Cu	mg/kg TS	3790	13200	7370	21700
Hg	mg/kg TS	1.51	2.33	2.61	
Mo	mg/kg TS	99.0	149	39.6	
Nb	mg/kg TS	4.47	2.00		
Ni	mg/kg TS	585	490	364	510
Pb	mg/kg TS	2970	3360	2600	4790
S	mg/kg TS	5160	7570	3940	4890
Sb	mg/kg TS	377	923	336	271
Sc	mg/kg TS	0.949	0.274	1.20	
Sn	mg/kg TS	419	488	208	266
Sr	mg/kg TS	408	453	205	218
V	mg/kg TS	89.1	34.6	39.1	
W	mg/kg TS	189	94.5	179	
Y	mg/kg TS	9.47	7.63	4.49	
Zn	mg/kg TS	37200	49700	24300	14600
Zr	mg/kg TS	553	366	244	142
Br	mg/kg TS	510	2670	436	1030

Cl	mg/kg TS	4420	4890	10200	11900
I	mg/kg TS	2.78	2.60	1.88	
Li (*)		0.0497	0.305		

*** Based on data from Nieminen et al. (2006)**

4. Life cycle inventories

4.1 Inventories for treatment processes

4.1.1 LCI of SR sorting

SR sorting is modelled based on data for landfilled SR retrieved from Møller et al. (2014), and then modified to account for the specific characteristics of newly produced SR. In fact, in comparison with the landfilled SR in Møller et al. (2014), the amounts of metals in the newly produced SR are considerably lower. Furthermore, the majority of the copper is supposedly contained in cables and thus cannot be sorted out without prior processing. It is thus assumed that sorting is performed on the following materials: >4 mm fraction for further processing, ferrous metals, non-ferrous (aluminium) metals, plastics and glass. The different steps are presented in Figure 6, while the amount of materials sent to further treatment is shown in Figure 7.

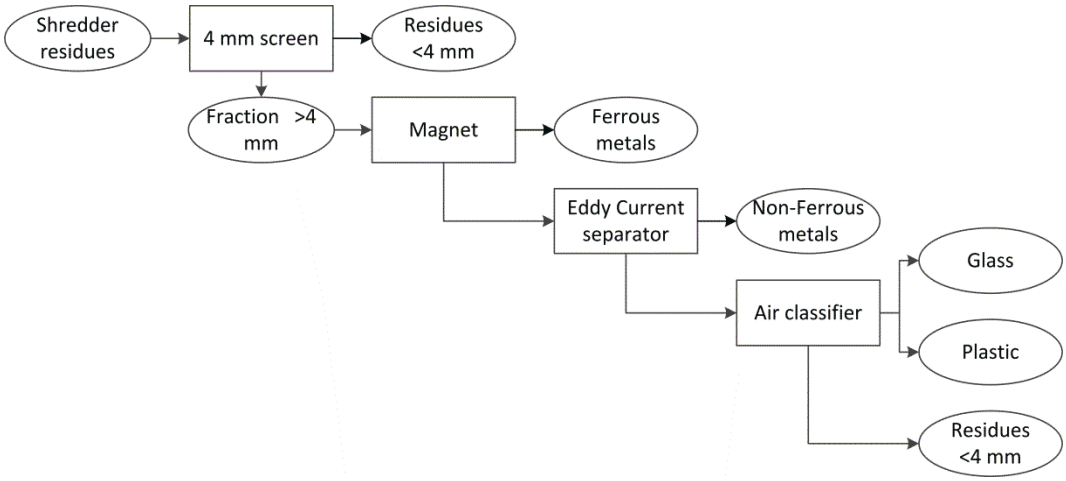


Figure 6 – Overview of the SR sorting process.

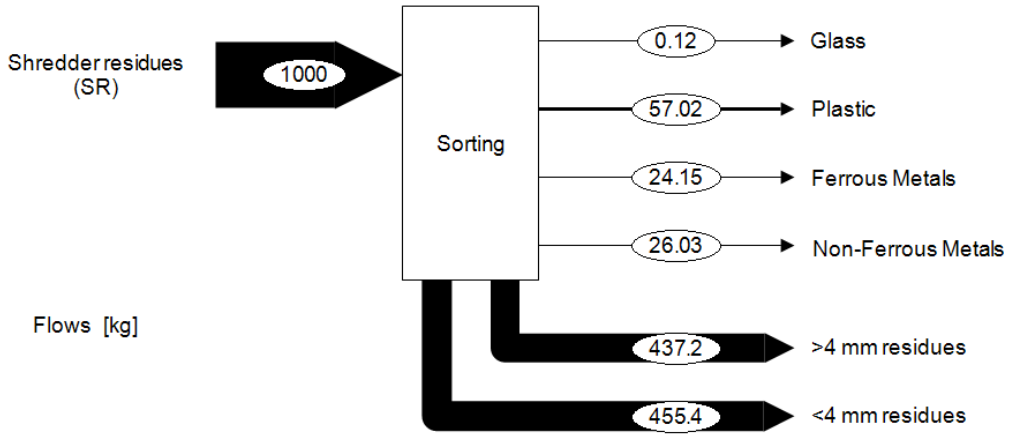


Figure 7 – Overview of material flows through the sorting facility.

The screen is assumed to sort 100% of the >4 mm fraction that will go on to the further processing steps explained in Chapter 3. As additional data on the efficiency of the process for sorting newly produced SR were not available, the dataset is thus based on information retrieved from Møller et al. (2014). As newly produced SR has a lower content of metals compared with disposed SR, average sorting efficiency is assumed to be similar to what was used in the sensitivity analysis in Møller et al. (2014). These average values are then combined with the higher values and a lower sorting efficiency of 50% to form a triangular distribution for sorting the different materials, as presented in Table 5. Data for energy consumption in the individual processing units are based on Møller et al. (2014) and shown in Table 6. The sorted materials are then sent for remanufacturing, where additional sorting to remove impurities can take place.

Table 5 – Efficiencies of individual processing units in sorting SR, expressed as a triangular distribution.

Process	Unit	Min	Mean	Max
Magnetic separator	% input	50	66	94
Eddy current separator	% input	50	66	92
Plastic seperation[1]	% input	16	48	80
Glass seperation[2]	% input	20	40	60

[1] Based on 80% sorting from Hansen et al. (2012), and an assumption of 20-60-100% of a quality to be recycled
[2] Based on 40% sorting from Hansen et al. (2012), and an assumption of +/-50% range

Table 6 – Input of energy to individual processing units in sorting SR.

Process	Input	Unit	Amount
4mm Screen	Electricity	kWh/kg throughput	0.0225
Magnetic separator	Electricity	kWh/kg throughput	0.0006
Eddy Current separator	Electricity	kWh/kg throughput	0.005
Plastic and glass separation	Electricity	kWh/kg throughput	0.005

4.1.2 LCI of SR combustion at a waste incinerator plant

Scenario 2 includes SR combustion at a waste incineration plant. In EASETECH, waste incineration plants are modelled by mapping emissions into the air of individual compounds and/or their transfer into other waste products. Individual compounds contained in waste are distributed into the air and as solid outputs using transfer coefficients (TCs). Air emissions from waste incineration are further divided into two types: input-specific emissions and process-specific. The former are proportional to the concentration of individual compounds in the waste and are thus modelled using TCs. This type of emission includes, for example, heavy metals and CO_2. Process-specific emissions are largely independent of the composition of the waste, and they are solely a function of the plant's operational characteristics. Examples of process-specific emissions include NOx, SO2 and dioxin emissions. In addition, the LCI for waste incineration includes internal energy consumption, and the consumption of auxiliaries is accounted for, together with the substitution value of the energy produced. In the present study, SR incineration is modelled similarly to how Møller et al. (2014) modelled excavated SR, and thus it is based on the inventory of a generic Danish waste incinerator described in Jacobsen et al. (2013).

4.1.2.1 Technology description

The incineration plant is modelled on the basis of Jacobsen et al. (2013), who provided a comprehensive inventory dataset for a state-of-the-art waste incineration plant in Denmark. This inventory is based on data collected from state-of-the-art Line 5 of Vestforbrænding I/S for 2011. Line 5 is equipped with a grate-type furnace, and its flue gas cleaning system includes wet scrubbing for acid gas removal, an SNCR deNO_x system and an activated carbon filter for dioxin and Hg removal. The dataset is based on both process- and input-specific emissions, depending on whether the emissions are mostly controlled by the plant operation or are mainly related to the content of specific compounds in the waste input. Leaching from bottom ash (i.e. slag), used for road

construction, is estimated using results from batch leaching test EN 12457-1, performed at L/S = 2 l/kg. The recipient of bottom ash leachate is supposed to be freshwater.

Energy contained in waste is recovered and used to produce electricity and hot water. The net energy recovery efficiencies are 22% for electricity generation and 73% for heat production, according to data reported in 'Technology Data for Energy Plants' (Energistyrelsen, 2012). The generated electricity is delivered to the grid, while hot water is delivered to a district heating network. Electricity is assumed to substitute for coal-based marginal electricity production, while heat is assumed to substitute for average Danish district heating, as explained in Chapter 4.2.

Bottom ash is treated to recover aluminium and iron. The total recovery depends on the efficiency of the sorting process and the degree of oxidation of the separated metal, where the latter is determined by the material thickness, as oxidation decreases with increasing thickness (Malmgren-Hansen et al., 2002). Because of the lack of specific data for SR, data for metal recovery from bottom ash generated during the combustion of general waste in a Danish incinerator were used. The recovery of iron was thus assumed to have an efficiency of 85% (Allegrini et al., 2014). The oxidation grade for >4 mm residues was assumed at 7.5%, as reported by Møller et al. (2014), and the recovery of non-oxidised aluminium was assumed to have an efficiency of 49%, calculated on the total input of Al into the combustion process, as reported in Biganzoli & Grosso (2013) and explained in Møller et al. (2014).

4.1.2.2 Inventory dataset

The inventory dataset for the combustion of SR at a waste incinerator in Denmark is presented in Table 7. As mentioned above, basic inventory data were taken from Jakobsen et al. (2013). This assumption is considered valid upon the condition that ASR is co-combusted with regular municipal solid waste up to a share of 12-14% and is made based on the following considerations:

- Astrup et al. (2011a) tested co-combustion by blending 14% ASR into municipal solid waste and reported that despite the content of individual compounds significantly increasing with the introduction of ASR into the feedstock blend (compared with municipal residual waste), emissions from the stack were not affected. In some cases, TCs into the air seemed eventually to decrease with the addition of ASR, possibly because a significant share of the metals is likely to be embedded in a non-combustible matrix and is thus not volatilised in the flue gas phase. This may suggest that TCs for ASR are somehow lower than for residual waste, and it can hence be conservatively assumed that TCs for ASR are equal to regular waste. While results are only available for As, Cd, Cr, Pb and SB, similar behaviour for other compounds is also assumed.
- Redin et al. (2001), Astrup et al. (2011a), Nedenskov (2011) and Vermeulen et al. (2012) reported regular functioning of the flue gas cleaning system and rather constant emissions during the co-combustion of ASR with municipal solid waste. Redin et al. (2001) tested 20% ASR, Astrup et al. (2011a) performed a co-combustion test with 14% ASR on a mass basis, while Nedenskov (2011) tested a mix with 12.4% ASR and Vermeulen et al. (2012) a mix with 25% and 39% ASR.
- The Danish EPA gave to I/S Reno-Nord environmental approval for the co-combustion of ASR with municipal solid waste up to 12.5% of mass (Seerup, 2012). In the approval, the Danish EPA assessed that the flue gas cleaning system at I/S Reno-Nord would ensure that the emission of heavy metals into the air would not rise, or at least not rise significantly, as a consequence of ASR co-combustion.

Table 7 – Inventory dataset for the combustion of 1 ton of SR at a waste incinerator.

Type	Parameter	Unit	Jakobsen et al. (2013)	Deviation
Input material	SR	Ton ww	1	
	NaOH	kg/ton ww	0.024	
	Activated carbon	kg/ton ww	1.04	
	CaCO3	kg/ton ww	5.67	
	NH3	kg/ton ww	1.53	
	Water	kg/ton ww	397	
	Ca(OH)2	kg/ton ww	0.34	
	Polymer	kg/ton ww	0.0006	
	HCl	kg/ton ww	0.0056	
	TMT	kg/ton ww	0.395	
Output	Electricity	% LHV, net	22	
	Heat	% LHV, net	73	
Process-specific air emission	HCl	kg/ton ww	0.0053	
	CO	kg/ton ww	0.033	
	NOx	kg/ton ww	0.849	
	HF	kg/ton ww	0.00039	
	Dioxin	kg/ton ww	$1.8*10^{-11}$	
	PM	kg/ton ww	0.003	
	SO2	kg/ton ww	0.00291	
Input-specific air emissions (% of content in waste input)	Cl	% input	0.1073	0.3036
	S	% input	0.099	0.2801
	As	% input	0.0121	0.0366
	Cd	% input	0.0064	0.0174
	Cr	% input	0.0394	0.1336
	Cu	% input	0.00261	0.0092
	Hg	% input	0.7476	2.0014
	Ni	% input	0.0329	0.1007
	Pb	% input	0.00081	0.0023
	Sb	% input	0.0119	0.0337
	Zn*	% input	0.0717	0.0994
Bottom ash composition (% of content in waste input)	Cl	% input	5.3	2.81
	S	% input	23.99	12.70
	As	% input	40.62	21.50
	Cd	% input	11.83	6.26
	Cr	% input	83.15	44.01
	Cu	% input	92.63	49.03
	Fe	% input	96.92	51.30
	Hg	% input	2.38	1.26
	Mo	% input	96.61	51.14
	Ni	% input	87.32	46.22
	Pb	% input	48.47	25.65
	Sb	% input	38.91	20.59
	Se	% input	22.38	11.85
	Zn	% input	51.76	27.40
Fly ash composition (% of content in waste input)	Cl	% input	32.13	13.69
	S	% input	60.91	25.95
	As	% input	58.92	25.10
	Cd	% input	88.13	37.55
	Cr	% input	16.77	7.15
	Cu	% input	7.35	3.13
	Fe	% input	3.06	1.30
	Hg	% input	96.25	41.01
	Mo	% input	2.54	1.08
	Ni	% input	12.56	5.35
	Pb	% input	51.29	21.85
	Sb	% input	59.84	25.50

	Se	% input	76.73	32.69
	Zn	% input	48.18	20.53
* Calculated based on Vermeulen et al. (2012)				

- Vermeulen et al. (2012) investigated the incineration of 100% ASR in a rotary kiln plant, reporting process-specific emissions very similar to Jakobsen et al. (2013). With regards to input-specific emissions, Vermeulen et al. (2012) reported slightly higher TCs for some of the assessed compounds. The small differences (e.g. Cd and Cr) may be associated with the fact that while Jakobsen et al. (2013) covered a grate furnace incineration plant, Vermeulen et al. (2012) performed a test on a rotary kiln process used for incinerating hazardous waste, and thus running on higher temperature.
- Nedenskov (2013) and Hyks & Astrup (2009) reported that the quality of solid residues originating from the co-combustion of SR with municipal solid waste (MSW) did not significantly differ when compared with bottom ash from incinerating 100% residual municipal solid waste.

To complement the data taken from Jakobsen et al. (2013), uncertainty associated with individual parameters was estimated (see Table 7, details in Appendix 3):

- For air emissions, using data from Vermeulen et al. (2012);
- For the composition of solid residues, using data from Mancini et al. (2014) and
- For leaching from bottom ash, using data from Nedenskov (2013).

4.1.3 LCI of pyrolysis of SR

Pyrolysis is an endothermic process during which the feedstock is heated under anaerobic conditions so that carbonaceous compounds therein are converted into energy-rich pyrolysis products such as oil, gas (mainly CO and H_2) and coke. As pyrolysis occurs in an oxygen-free environment and at a lower temperature than combustion (from 400°C to 900°C), metals eventually present in the feedstock are not oxidised. A detailed description of SR pyrolysis is provided by Høstgaard et al. (2012) based on small-scale laboratory tests.

Since the pyrolysis process is endothermic, its operation requires the input of energy. This energy is typically obtained by using the pyrolysis gas for heating the reactor, while oil and coke can be employed for energy production to be exported. Pyrolysis gas, oil and coke can – in principle – be used as fuels in a number of technologies, while in reality their application is limited because they carry undesirable levels of e.g. heavy metals (Høstgaard et al., 2012). The present study thus assumes that the pyrolysis products are utilised in a waste incineration plant, as its feasibility is reported in Høstgaard et al. (2012). The inventory for the incineration of pyrolysis products is modelled according to the dataset presented in Chapter 4.1.2.

4.1.3.1 Technology description

An average pyrolysis process for treating SR was obtained based on physicochemical compositions of SR input and pyrolysis products (i.e. char, oil and gas) retrieved from a number of literature studies. Data were reconciled to balance mass, energy and substance flows simultaneously while estimating the uncertainty of the system. The mass and energy balances of an average pyrolysis process for the treatment of SR are shown in Figure 8 and Figure 9, while additional charts for individual compounds can be found in Appendix 4, together with a detailed description of the reconciliation process. The recovery of metals from char is possible because the organic part of the char is typically porous and can be screened out rather easily (Møller et al., 2014). While forcing data reconciliation based on materials compositions (i.e. SR input, char, oil and gas), in the mass balance calculation, sorting efficiency was assumed at between 80% and 100%.

Figure 8 and Figure 9 indicate that mass and energy recoveries have rather different patterns, with char being the main material output, while oil carries the largest energy flow. In addition, it is evident that all flows are associated with significant uncertainty, possibly because the displayed

average process was built on a rather large set of data, originating from different test conditions. This is indeed a consequence of limited data availability and lack of full-scale experiments on representative Danish SR.

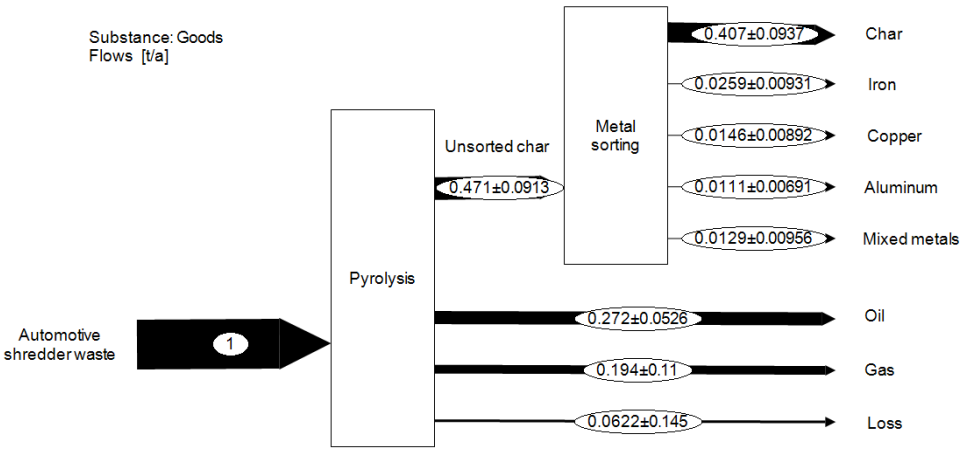


Figure 8 – Mass balance for the pyrolysis of 1 ton of SR.

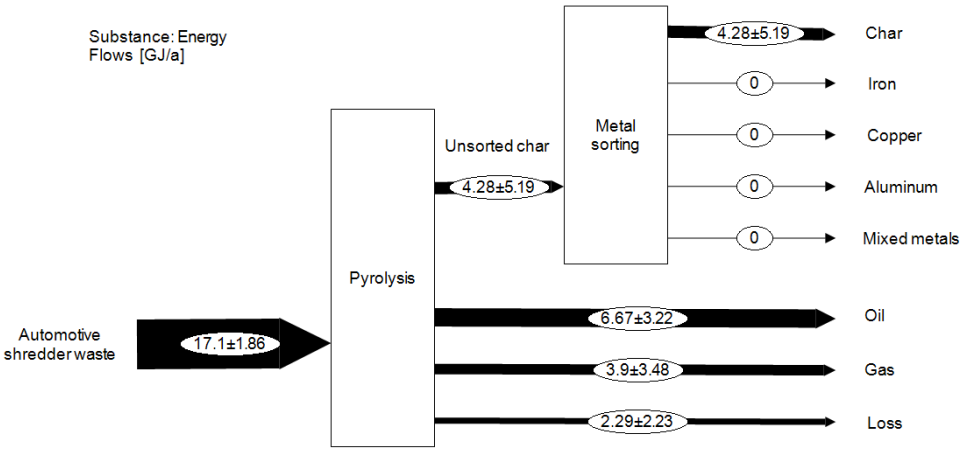


Figure 9 – Reconciled distribution of energy during the pyrolysis of 1 ton of SR.

4.1.3.2 Inventory dataset

Mass, energy and substance reconciliation allowed for calculating process TCs for individual outputs. An overview of the inventory datasets used in the LCA modelling is presented in Table 8, together with the uncertainty associated with individual parameters. A detailed description of the inventory is provided in Appendix 4.

4.1.4 LCI of disposal of SR in landfill

The inventory for landfilling SR was developed similarly to how it was applied by Møller et al. (2014). Gas emissions from the depositing of SR into landfill sites were estimated based on experimental results reported in Scheutz et al. (2010a, 2010b, and 2011). Emissions related to leachate production were estimated based on the modelling of experimental results reported in Hansen et al. (2011b) and the estimation of uncertainty was based on data from Hansen et al. (2011a). An overview of the inventory dataset is provided in Table 9, while additional details about the inventory creation are provided in Appendix 5. In addition, electricity consumption of 97 kWh/ton_{SR} was considered for leachate treatment at a wastewater treatment plant (Møller et al., 2014).

Table 8 – Inventory dataset for the pyrolysis of SR.

Type	Parameter	Unit	Mean	St. Dev.
Input	SR	ton	1	
	Electricity	MJ/ton ww	248	
	Pyrolysis oil	GJ/ton ww	0.29	
	Pyrolysis gas	GJ/ton ww	0.21	
Output	Char	% LHV, net	25.0	30.1
	Oil	% LHV, net	38.9	18.3
	Gas	% LHV, net	22.7	20.1
	Ferrous metals	kg/ton ww (% input)	25.9 (37.0)	9.31 (12.7)
	Non-Ferrous metals	kg/ton ww (% input)	25.7 (8.24)	11.3 (6.18)
	Unsorted metals	kg/ton ww (% input)	12.9 (6.73)	9.56 (7.70)
Transfer to char (% of content in waste input)	Al	% input	17.9	10.1
	C	% input	29.2	10.5
	Ca	% input	66.1	16.8
	Cd	% input	90.3	109
	Cl	% input	71.1	81.4
	Cr	% input	26.0	30.4
	Cu	% input	23.9	16.2
	Fe	% input	47.4	14.2
	Hg	% input	0.75	0.71
	Mn	% input	61.3	19.1
	N	% input	39.8	17.6
	Ni	% input	32.0	47.2
	Pb	% input	34.2	38.4
	S	% input	72.4	11.5
	V	% input	21.0	12.8
	Zn	% input	14.8	7.30
Transfer to oil (% of content in waste input)	Al	% input	0.008	0.005
	C	% input	59.0	9.51
	Ca	% input	0.181	0.313
	Cd	% input	0.048	0.032
	Cl	% input	2.74	1.61
	Cr	% input	0.164	0.089
	Cu	% input	0.002	0.002
	Fe	% input	0.091	0.158
	Hg	% input	3.88	0.749
	Mn	% input	0.008	0.002
	N	% input	58.0	18.3
	Ni	% input	0.180	0.202
	Pb	% input	0.053	0.073
	S	% input	20.8	12.0
	V	% input	0.027	0.031
	Zn	% input	0.032	0.024
Transfer to gas (% of content in waste input)	Al	% input	0.994	1.00
	C	% input	9.34	2.34
	Ca	% input	0.564	0.321
	Cd	% input	0.100	0.100
	Cl	% input	1.000	1.00
	Cr	% input	0.100	0.100
	Cu	% input	0.100	0.100
	Fe	% input	0.100	0.100
	Hg	% input	94.6	1.41
	Mn	% input	0.100	0.100
	N	% input	0.000	0.000
	Ni	% input	5.00	20.0

	Pb	% input	0.100	0.100
	S	% input	0.000	0.000
	V	% input	0.100	0.100
	Zn	% input	0.100	0.100

Table 9 – Inventory dataset for the disposal of SR in a landfill (after Møller et al., 2014).

Type	Substance	Production	Release*	Uncertainty (%)
Gas		**kg/kg$_{SR}$**	**kg/kg$_{SR}$**	
	CH4	$1.2*10^{-2}$	$8.4*10^{-3}$	
	CFC-11	$1.3*10^{-7}$	$1.3*10^{-8}$	
	HCFC-21	$1.8*10^{-6}$	$8.8*10^{-7}$	
	HCFC-31	$1.3*10^{-6}$	$9.1*10^{-7}$	
	HFC-41	$6.5*10^{-8}$	$6.5*10^{-8}$	
	CFC-12	$6.5*10^{-8}$	$4.6*10^{-8}$	
	HCFC-22	$1.3*10^{-7}$	$1.0*10^{-7}$	
	HFC-32	0.0	0.0	
	HFC-134a	$6.5*10^{-7}$	$6.5*10^{-7}$	
	HCFC-141b	$1.3*10^{-7}$	$1.0*10^{-7}$	
Leachate		**kg/kg$_{SR}$**	**% of leachate content**	
	Total-N	$4.5*10^{-4}$	9.80%	
	Total-P	$5.5*10^{-6}$	9.00%	
	Cl^-	$9.8*10^{-4}$	100.00%	60
	F^-	$4.0*10^{-6}$	14%	38
	SO_4^{2-}	$2.2*10^{-5}$	14%	39
	HCO_3^-	$7.3*10^{-3}$	14%	
	NVOC (DOC)	$1.2*10^{-3}$	14%	
	$S-S^{2-}$	$6.6*10^{-7}$	14%	
	Al	$1.1*10^{-7}$	0.30%	48
	As	$1.0*10^{-7}$	87.00%	16
	Ba	$1.8*10^{-6}$	13.90%	29
	Ca	$5.3*10^{-4}$	92.10%	39
	Cd	$7.1*10^{-11}$	13.90%	74
	Co	$2.0*10^{-9}$	17.00%	37
	Cr (VI)	$2.3*10^{-9}$	11.20%	
	Cr (tot)	$2.0*10^{-8}$	11.20%	68
	Cu	$7.0*10^{-9}$	2.00%	23
	Fe	$4.1*10^{-5}$	1.70%	93
	Hg	$1.7*10^{-10}$	10.30%	131
	K	$4.5*10^{-4}$	94.30%	19
	Mg	$7.0*10^{-4}$	95.10%	26
	Mn	$1.3*10^{-6}$	47.10%	34
	Mo	$9.6*10^{-9}$	95.00%	31
	Na	$8.5*10^{-4}$	89.20%	44
	Ni	$1.2*10^{-7}$	47.10%	47
	Pb	$7.1*10^{-10}$	2.70%	71
	Sb	$6.7*10^{-9}$	45.80%	34
	Se	$2.3*10^{-9}$	14%	2
	Si	$4.3*10^{-5}$	14%	45
	V	$3.1*10^{-8}$	16.90%	68
	Zn	$9.6*10^{-8}$	7.30%	61
	Benzene (sum)	$-3*10^{-7}$	30%	
	PAH	$-6.0*10^{-11}$	29%	
	PAH	$-6.0*10^{-11}$	29%	
* For gas: release after oxidation; for leachate: release after wastewater treatment.				

4.1.5 LCI of co-combustion of SR in a cement kiln

As data regarding the co-combustion of SR in a cement kiln are not available for Denmark, a generic process was assumed for LCI modelling. Data on SR composition and emissions released into the air were retrieved from Vermeulen et al. (2012) and used to estimate TCs emitted into the air for a range of compounds. This project was chosen because it was the only available published study providing both the composition of SR input into a cement kiln as well as emissions into the air of a number of compounds, thereby allowing us to determine air emissions as input-specific. With regards to solid residues, it was considered that the inert part of SR (i.e. ash and metals) would end up in the clinker, and thus no further modelling was included. The inventory dataset for the co-combustion of SR in a cement kiln is presented in Table 10.

Table 10 – TCs (%) into the air for the co-combustion of SR in a cement kiln.

	TCs (%) into air	
Compound	**Median**	**Deviation**
C	100	
S	0.0397	0.1071
As	0.0001	0.0003
Cd	0.0172	0.0478
Cr	0.0005	0.0013
Cu	0.0273	0.0790
Hg	0.0367	0.1038
Ni	0.0022	0.0089
Pb	0.0571	0.2195
Zn	0.3730	1.2975

4.1.5.1 Feasibility of the co-combustion of SR in a cement kiln

Cement kilns typically have restrictions on the quality of the fuel used for producing clinker. Co-combustion of SR may thus be questionable, if at all possible or allowed. The technical feasibility of the scenario was therefore assessed with some desktop-based considerations. The estimated composition of SR used for co-combustion in Scenario 4 was compared with acceptance values for RDF used in cement production at Aalborg Portland. The results of the comparison are showed in Table 11, where it is evident that for a large number of parameters quality acceptance criteria would not be met by SR with a composition as modelled in this study. This means that, in practice, the co-combustion of SR in a cement kiln would only be feasible for a 'designed' composition of SR, where, for example, metals are sorted out to a larger extent than is currently done. This conclusion, however, should be supported by additional tests.

Table 11 – Comparison between quality acceptance criteria for the co-combustion of RDF in a cement kiln (Aalborg Portland) and the compositing of SR going to a cement kiln as estimated for Scenario 4. Values marked in grey exceed the quality criteria.

Name of Fuel	**RDF – Refuse Derived Fuel**					
Origin	A solid recovered fuel (SRF/RDF), made from combustible non-hazardous waste (municipal solid waste, specific waste, industrial waste and commercial waste), mainly consisting of organic components such as Cl-free plastics and biodegradable waste.					
Physical appearance	Fine, fluffy product.					
Physical cleanliness	Shall be free from foreign materials such as metals, glass, PVC, oils etc., and shall not be more contaminated by salts, soils, heavy metals and liquids, as is normal for this type of material.					
Parameter	**Unit**	**Typical value**	**Lower**	**Upper**	**Standards**	**LCI Modelling**
LHV	kcal/kg (DB)	5,000	4,000		CEN/TS 15400	
LHV	MJ/kg (DB)	21	17		CEN/TS 15400	18.3
LHV	MJ/kg (AR)	17	16		CEN/TS 15400	
Moisture	% (AR)	10		15	CEN/TS 15414	9.57
Ash	% (AR)	<10		15	CEN/TS 14775	32.7
S	% (DB)			1	CEN/TS 15408	0.42
Cl	% (DB)	0.5		0.8	CEN/TS 15408	1.09
F	% (DB)			0.5	CEN/TS 15408	
Br	% (DB)			0.25	CEN/TS 15408	0.05
I	% (DB)			0.25	CEN/TS 15408	
Biomass Content	%-w (DB)	50				
Size, Length/Width	mm	25		40	CEN/TS 15415	
Size, Thickness	mm	< 3		5	CEN/TS 15415	
Metals						
Mercury (Hg)	mg/kg (DB)	<1		1	CEN/TS 15411	2.79
Cadmium (Cd)	mg/kg (DB)	<9		9	CEN/TS 15411	10.4
Thallium (Tl)	mg/kg (DB)	<1		1	CEN/TS 15411	
Total Group II Metals	mg/kg (DB)	<10		10		
Antimony (An)	mg/kg (DB)	<150		150	CEN/TS 15411	359
Arsenic (As)	mg/kg (DB)	<20		20	CEN/TS 15411	52.6
Chromium (Cr)	mg/kg (DB)	<150		150	CEN/TS 15411	418
Cobalt (Co)	mg/kg (DB)	<20		20	CEN/TS 15411	
Copper (Cu)	mg/kg (DB)	<500		500	CEN/TS 15411	7887
Lead (Pb)	mg/kg (DB)	<200		200	CEN/TS 15411	2787
Manganese (Mn)	mg/kg (DB)	<150		150	CEN/TS 15411	1115
Nickel (Ni)	mg/kg (DB)	<70		70	CEN/TS 15411	390
Tin	mg/kg (DB)	<15		15	CEN/TS 15411	222
Vanadium (V)	mg/kg (DB)	<10		10	CEN/TS 15411	41.8
Total Group III Metals	mg/kg (DB)	<800		800		
Organics	Must not contain dioxins, furans, PCBs or other hazardous components.					
Waste Code	191210 according to the European Waste Catalogue (EWC).					
Quality Control	The supplier must have a quality control system and provide a declared analysis of LHV, Moisture, Ash, S/Cl/F/Br/I & Heavy Metals.					
DB = Dry Basis; AR = As received						

4.1.6 LCI of the biological treatment of SR before disposal in a landfill

As described in Chapter 3, all the assessed scenarios include the landfilling of a significant amount of unsorted residues (i.e. residues <4 mm in Scenarios 2, 3 and 4, and both <4 mm and >4 mm residues in Scenario 1). To prevent both landfill gas formation and emissions of other potentially harmful substances into the environment, aerobic biological treatment is employed to reduce the organic content and thus stabilise the fine residues. This treatment will also ensure compliance with quality criteria for waste landfilling (Møller et al., 2014), as specified in EU Council Decision 2003/33/EC (European Council, 2003).

A comprehensive description of the process involved in the biological treatment of SR fractions to be landfilled is reported in Møller et al. (2014). Inventory data are provided by BIOSA Denmark Aps, based on information originating from an SR-treating plant located in Germersheim, Germany.

4.1.6.1 Technology description

The biological process for stabilising SR fractions to be landfilled consists of an ad hoc modified composting process. SR is mixed with water and a bacterial suspension and then laid in up to 100 m long composting windrows. The composting process lasts for 8-10 weeks at temperatures in the range 55-75˚C. The composting windrows are turned periodically using windrow turners. As a result of the composting process, mass is significantly reduced, while the stabilised SR is darker and presents a finer grain size compared with the SR input into the process. According to chemical analyses performed in Germersheim, the treated SR material complies with the limits for waste to be landfilled.

As mentioned previously, the process makes use of inoculum, consisting of a bacterial suspension, which is meant at enhancing the SR degradation process. However, as the effects of inoculum addition cannot be quantified, the biological treatment of SR is hereby modelled as a conventional windrow composting process (i.e. without the addition of microorganisms). This exclusion is expected not to have a significant impact on the results, for two reasons. Firstly, the use of bacterial suspension is rather minimal (see Table 12), while other studies (e.g. Boldrin et al., 2013) including even larger amounts of enzyme-based inoculum have concluded that enzyme production is not a main contributor to total impact. Secondly, a main effect of inoculum utilisation would be a shortening of the retention time during the composting process and a subsequent reduction in electricity and fuel consumption for process operations. However, as shown in Table 12, the consumption of diesel and electricity is rather minor, and results on potential impacts, presented in Chapter 5, will show that these make a rather non-significant contribution.

4.1.6.2 Inventory

As there is currently no operating plant for the biological treatment of SR in Denmark, the inventory dataset is built on data provided by BIOSA Denmark Aps, based on information originating from a plant treating SR and located in Germersheim, Germany. The information includes mass balance and data describing the biodegradation and emissions occurring during the treatment process, as well as emissions from the process. Data on energy consumptions were estimated by Møller et al. (2014) based on literature data. An overview of the inventory dataset used for modelling the biological treatment of SR is provided in Table 12.

4.2 Marginal energy production

The treatment of increasing amounts of SR will result in increasing demand for energy (in particular electricity) to operate the treatment processes. On the other hand, when SR is thermally treated and energy is recovered, the production of electricity and district heating also has an effect on the energy market, because the increasing production of electricity and heat from incineration displaces energy production somewhere else in the system. The substituted marginal electricity was presumed to by coal, as reported in Astrup et al. (2011b). To cover the future potential development

of the Danish electricity system, gas-based and wind-based marginal electricity scenarios were also modelled. Marginal heat is much dependent on the settings of the local district heating system. As described in Jacobsen et al. (2013), it was assumed that the heat from SR incineration substitutes for average Danish district heating. Inventories for marginal energy productions are provided in Appendix 7.

4.3 Selection of datasets for the primary production and recycling of materials

Inventory datasets for the primary production and recycling of materials were selected according to the recommendations made by Møller et al. (2014). An overview of the datasets used herein is provided in Table 13, where it is evident that data were retrieved from two sources, namely the EASETECH and the Ecoinvent databases. As explained in Møller et al. (2014), these data were selected by prioritising recently developed datasets as well as data having a geographical scope in accordance with the scope of the present study. A detailed description of the dataset for plastic recycling is provided in Appendix 7, while a description of the other datasets found in Table 13 is provided in Møller et al. (2014).

Table 12 – Inventory dataset for the biological treatment of SR (adapted from Møller et al., 2014).

Type	Parameter	Unit	Quantity
Input	SR	ton	1
	Bacterial suspension	l/ton	3
	Electricity	kWh/ton	0.2
	Diesel	l/ton	3
Process	C degradation	% of biogenic C	17.5
	N degradation	% of organic N	2.5
Emissions	CH_4	% of degraded biogenic C	2.7
	N_2O	% of degraded organic N	22
	NH_3	% of degraded organic N	78
	TOC	g/ton	17-19

Table 13 – Overview of inventory datasets for the primary production and recycling of materials used in the LCI modelling of SR treatment, as selected by Møller et al. (2014).

Material	Process name	Type	Database	Year	Reference
Aluminium	Aluminium, Al (Primary), World average, 2005	Primary	EASETECH	2005	Clavreul et al., 2014
	Aluminium scrap to new alu. sheets (re-melting), Sweden, 2007	Recycling	EASETECH	2007	Clavreul et al., 2014
Iron	Steel sheets (97.75% primary), Sweden, 2008	Primary	EASETECH	2008	Clavreul et al., 2014
	Shredding and reprocessing of steel scrap, Sweden, 2007	Recycling	EASETECH	2007	Clavreul et al., 2014
Copper	Copper, primary, at refinery/RER U	Primary	Ecoinvent	1994	Classen et al., 2009
	Copper, secondary, at refinery/RER U	Recycling	Ecoinvent	1994	Classen et al., 2009
Plastic	Polyethylene, HDPE, granulate, at plant/RER S	Primary	Ecoinvent	1999	Hischier, 2007
	Postconsumer recycled HDPE pellet, US, 2011	Recycling	EASETECH	2011	Franklin Associates, 2011
Glass	Glass – bottle (primary), EDIP, 1990	Primary	EASETECH	1990	Miljøstyrelsen, 1998
	Glass cullet to new products (re-melting), EDIP, 1990	Recycling	EASETECH	1998	Miljøstyrelsen, 1998

4.4 Quality Indicator Values for processes used in the LCA

To obtain a more quantitative expression of process quality, all processes used in the study were assigned a quality indicator value, for the external processes of primary production and recycling, as well as processes specifically designed for this LCA. The values were assigned to five indicator categories, as seen in Table 14, based on Weidema & Wesnæs (1996) and as described in Chapter 2.10. They cover the following categories: 'credibility', 'completeness' and 'temporal, geographical and technological correlation'. Assigned indicator values are computed by comparing the processes with the processes as they should be, in order to fit fully into the environmental assessment scenarios.

Table 14 - Scoring of processes for use in the environmental assessment. All processes are scored with a value 1-5 ('1' is the best) for each of the quality indicators.

Material	Type	Process	Credibility	Completeness	Temporal Correlation	Geographical Correlation	Technological Correlation	Average
Processes designed specifically for this project								
SR	Unsorted	Sorting	3.6	3	1	1	3	2
	>4 mm	Incineration	1	3	2	1	3	2
	>4 mm	Pyrolysis	3	3	2	2	4	2.8
	>4 mm	Cement Kiln	4	4	2	4	4	3.6
	<4 and >4 mm	Biological Treatment	3	4	1	3	4	3
	<4 and >4 mm	Landfill	2	4	2	1	3	2.4
Process name in database								
Aluminium	Secondary	Aluminium scrap to new alu. sheets (re-melting), Sweden, 2007	1	4	3	3	2	2.6
	Primary	Aluminium, Al (Primary), World average, 2010	1	3	3	2	1	2.0
Iron	Secondary	Shredding and reprocessing of steel scrap, Sweden, 2007	1	4	3	3	2	2.6
	Primary	Steel sheets (97.75% primary), Sweden, 2008	1	3	3	3	2	2.4
Copper	Secondary	Copper, secondary, at refinery/RER U	2	2	3	2	2	2.2
	Primary	Copper, primary, at refinery/RER U	2	2	3	2	2	2.2
Plastic	Secondary	Plastic to HDPE granulate, Franklin Associate, USA, 2011	1	3	2	3	2	2.2
	Primary	Polyethylene, HDPE, granulate, at plant/RER S	1	1	4	1	2	1.8
Glass	Secondary	Glass cullet to new bottles (re-melting), Denmark, 1998	1	3	4	1	2	2.2
	Primary	Glass bottle, primary, EDIP, Denmark, 1990	3	3	4	1	1	2.8

5. Evaluation of potential environmental impacts

In this chapter, normalised results regarding potential environmental impacts from the treatment of SR in Denmark are presented:

- as a total impact for individual scenarios;
- disaggregated for individual processes within scenarios and
- disaggregated for individual material fractions.

Numerically-negative values represent avoided environmental impacts – i.e. environmental savings related to avoiding the production of material and energy – while numerically-positive values represent net impacts (i.e. burdens) on the environment. Characterised potential impacts are provided in Appendix 8.

5.1 Total potential environmental impacts

Potential impacts on non-toxic categories for the analysed scenarios are presented comparatively in Figure 10. The handling of SR according to Scenario 1 shows overall savings for Global Warming, Photochemical Oxidant Formation and Terrestrial Eutrophication, mostly because of the benefits associated with recovering and recycling plastic and aluminium fractions. For all impact categories, and within the uncertainty estimated in Figure 11, scenarios with increased energy recovery (i.e. S2-incineration and S4-cement kiln) from SR >4 mm show significantly better environmental performance compared with the baseline landfill-based scenario (i.e. S1). The increased savings are related to the avoided production of marginal energy (for S2) and coal (for S4), indicating that increased energy recovery is a desirable option for the non-toxic categories.

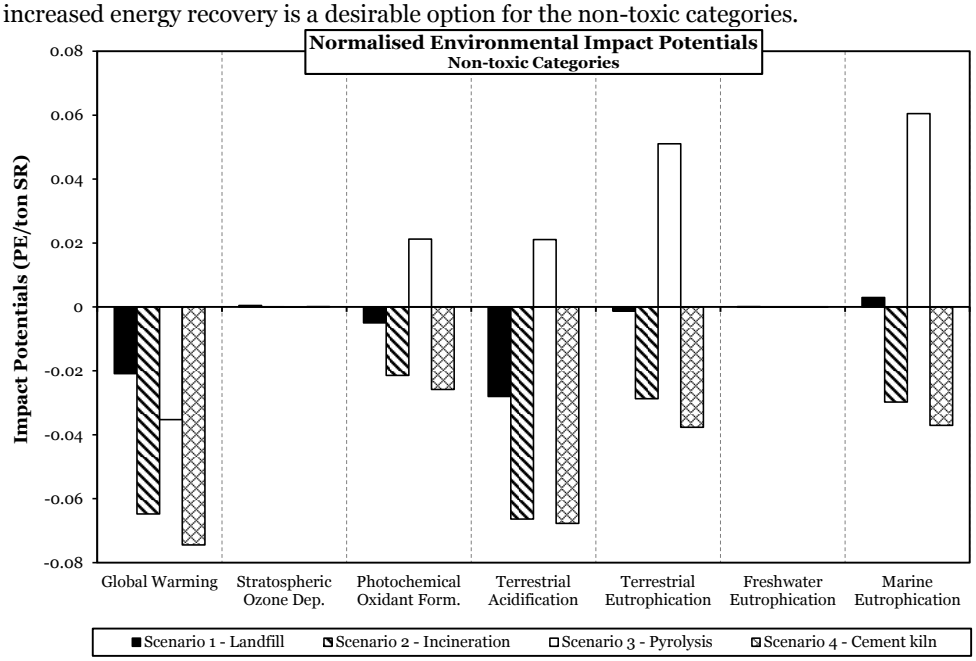


Figure 10 - Normalised potential non-toxic impacts from the treatment of 1 ton of SR in the four analysed scenarios (PE = person equivalent).

The use of pyrolysis as a thermal treatment option for the >4 mm fraction (i.e. S3) provides benefits for the Global Warming impact category, while resulting in significant environmental burdens in the remaining impact categories (in particular Photochemical Oxidant Formation, Terrestrial Acidification, Terrestrial Eutrophication and Marine Eutrophication), owing in particular to increased emissions of NO_x from the combustion of pyrolysis oil. However, when including the uncertainty (i.e. Figure 11), it is noticeable that results for S3 are associated with significant uncertainty, mostly owing to the fact that the dataset for SR pyrolysis included rather broad ranges of data (see Chapter 4.1.3). This suggests that a clear conclusion on the environmental sustainability of pyrolysis as a treatment technology for SR is not possible with currently available data and understanding of the process. This means that further experimental investigations are needed before it can be concluded on whether pyrolysis is a possible treatment technology for SR.

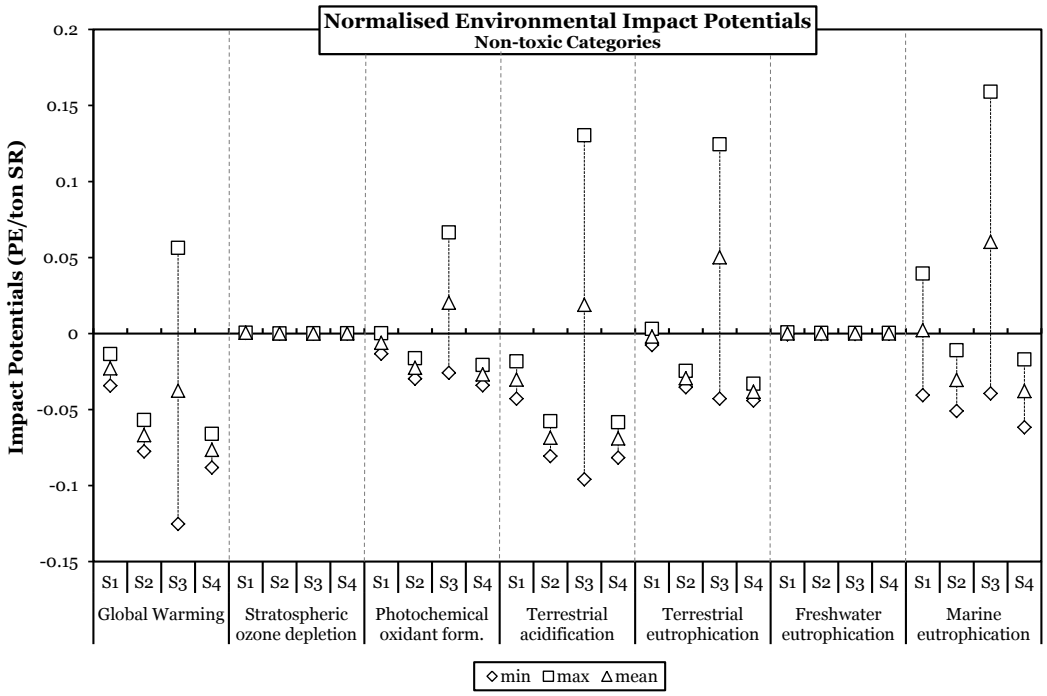


Figure 11 - Normalised potential non-toxic impacts from the treatment of 1 ton of SR in the four analysed scenarios (PE = person equivalent), including min-max range.

Results for the potential toxic impacts are shown in Figure 12, where it is evident that the four analysed scenarios have comparable profiles for the impact categories Human Toxicity Carcinogenic, Particulate Matter and Ionizing Radiation. The three scenarios including the thermal treatment of >4 mm fractions all show increasing potential impacts on Human Toxicity non-Carcinogenic and Ecotoxicity compared with the baseline scenarios, owing to increases in emissions into the air, specifically As and Hg. In particular, Scenario 4 presents potential impacts on Human Toxicity non-Carcinogenic and Ecotoxicity, which are far larger than the other scenarios, owing particularly to emissions into the air of Cu, Hg and Zn. Compared to other technologies, these large impacts are connected with greater emissions, owing to less efficient flue gas cleaning at the cement plant and higher volatility of these metals due to higher temperature in the kiln. Such findings confirmed the need for a pre-treatment step to produce an SR feedstock for the cement kiln with an RDF-like composition. This should especially include the additional screening and sorting of metals, to reduce the content of heavy metals in the feedstock and thereby significantly decrease emissions of compounds such as Cu, Hg and Zn into the air.

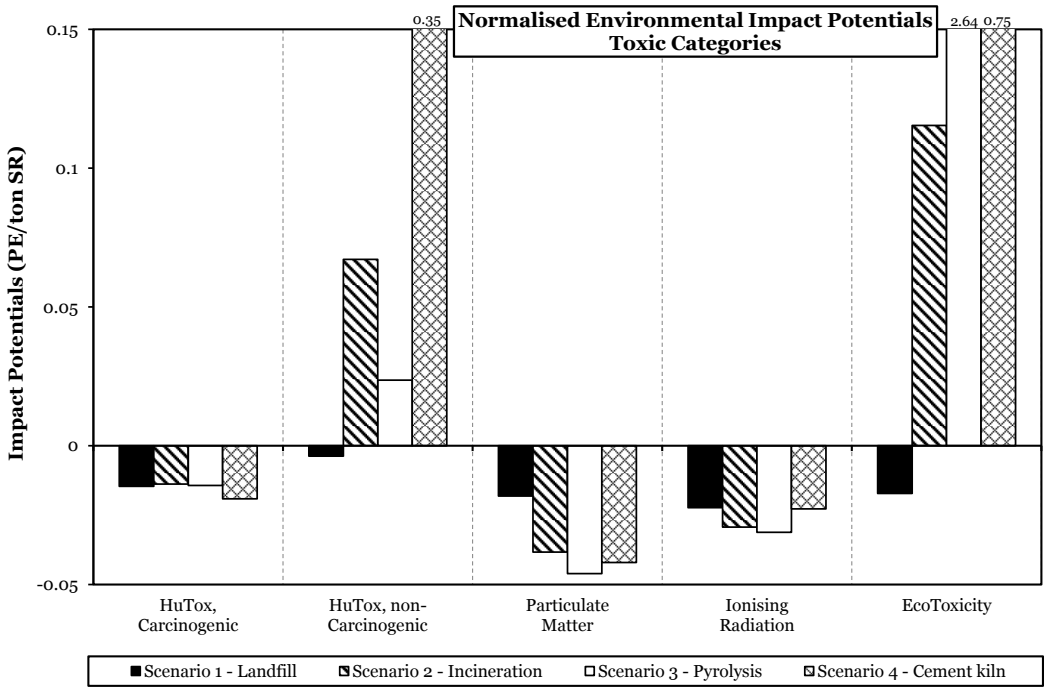


Figure 12 - Normalised potential toxic impacts from the treatment of 1 ton of SR in the four analysed scenarios (PE = person equivalent).

Uncertainty related to the potential impacts on toxic categories is visualised in Figure 13, where it is noticeable that the results for Human Toxicity non-Carcinogenic (S2 and S4) and Ecotoxicity (S3 and S4) are associated with very significant uncertainty. This is specifically due to large variabilities relating to some of the air emissions occurring during the different thermal treatment technologies (i.e. incineration for S2, pyrolysis for S3 and cement kiln for S4), as seen in Chapter 4.1.

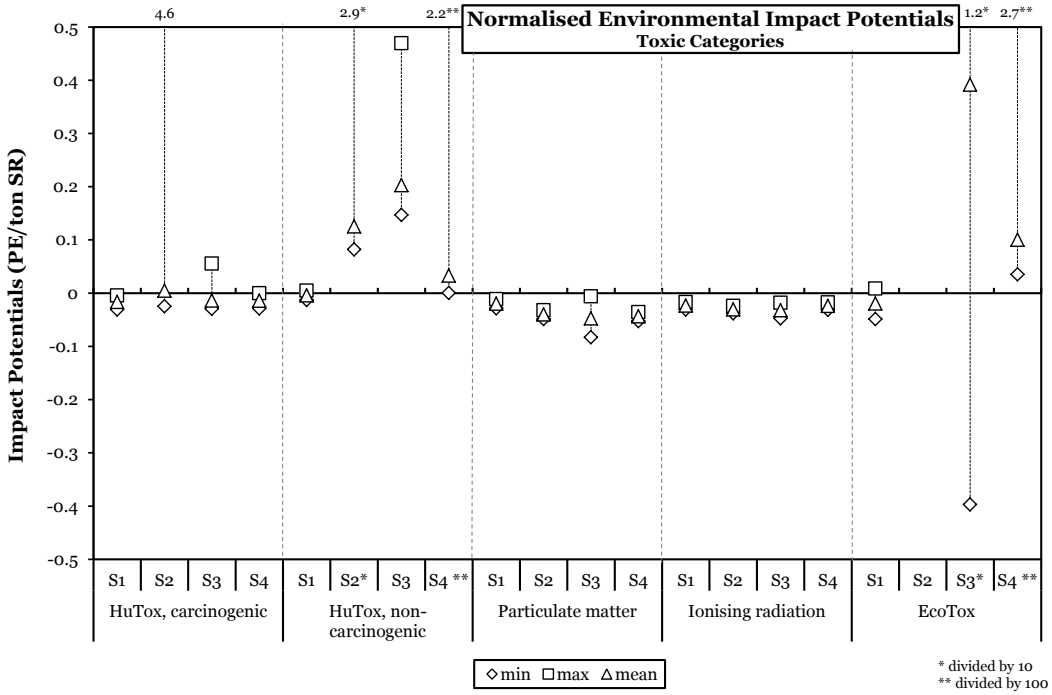


Figure 13 - Normalised potential toxic impacts from the treatment of 1 ton of SR in the four analysed scenarios (PE = person equivalent), including min-max range.

Figure 14 and Figure 15 show results from potential Resource Depletion. It is seen that, within the estimated uncertainty, scenarios including the thermal treament of the SR fraction >4 mm provide

increased savings in fossil-based Resource Depletion, as a consequence of increased energy recovery and the subsequent replacement of marginal energy production. With regards to mineral Resource Depletion, S3 is the only scenario showing significant savings, mostly owing to the recovery of copper from the SR char produced by pyrolysis.

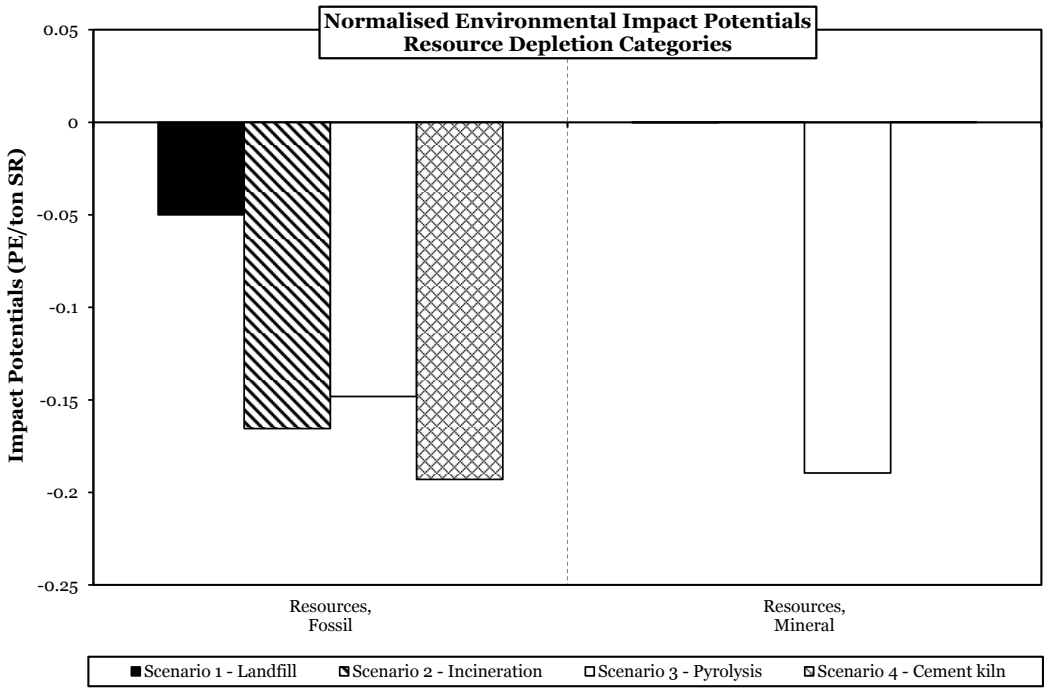


Figure 14 - Normalised potential Resource Depletion from the treatment of 1 ton of SR in the four analysed scenarios (PE = person equivalent).

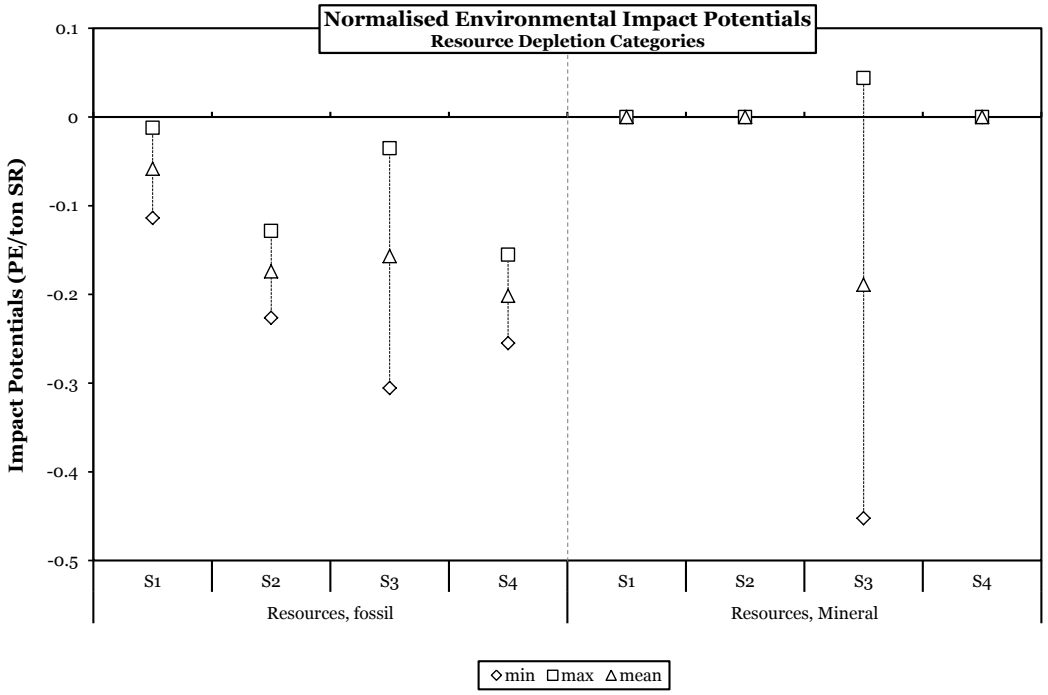


Figure 15 - Normalised potential Resource Depletion from the treatment of 1 ton of SR in the four analysed scenarios (PE = person equivalent), including min-max range.

5.1.1 Conclusions regarding total potential environmental impacts

When comparing the four analysed scenarios, the results indicate that introducing the thermal treatment of SR >4 mm results in significant improvements in the non-toxic impact categories.

Clear conclusions cannot be drawn with regards to the implementation of S3, as data available for SR pyrolysis are rather uncertain. Addition experimental investigation is hence needed, preferably at full-scale testing facilities.

The co-combustion of >4 mm fractions in a cement kiln could result in significant potential impacts on some of the toxic categories, owing in particular to some of the air emissions. These impacts could possibly be limited if SR were treated in a way that the content of some critical heavy metals (i.e. Cu, Hg and Zn) was reduced prior to feeding into the cement kiln.

5.2 Potential environmental impacts by process

In this chapter, potential environmental impacts are disaggregated according to individual processes, in order to identify and highlight the most sensitive ones.

5.2.1 Scenario 1

Disaggregated results for Scenario 1 are shown in Figure 16. In general, it is evident that the most significant contributions are associated with the material recycling of plastic and aluminium. Landfilling of <4 and >4 mm residues contributes potential impacts to Global Warming, Marine Eutrophication and Ecotoxicity, while bio-treatment prior to landfill contributes almost uniquely to fossil Resource Depletion, in connection with the use of energy expended to operate the process.

Estimated benefits are associated with avoiding emissions connected with the replacement of primary productions of plastic and aluminium, namely NMVOC, NO_x and SO_x (for Photochemical Oxidant formation), SO_x and some NO_x (for Acidification), NO_x and ammonia (for Terrestrial Eutrophication) and NO_x (for Marine Eutrophication). Regarding human toxicity categories, savings are due to avoiding emissions of Cr^{VI} during plastic production (for Human Toxicity, Carcinogenic) and Hg from aluminium production (for Human Toxicity, non-Carcinogenic). Emissions of Cr^{VI} and Hg are mainly related to the production and use of energy in the products, and not directly from the production itself. Savings in Ecotoxicity and Particulate Matter are mainly caused by recycling plastic and aluminium, and specifically in relation to reduced emissions of V and Cr^{VI} (Ecotoxicity) and PM and SO_x (Particulate Matter). The cause is here again contributed to by the use of energy in the production of materials.

With regards to burdens (i.e. positive impacts), significant contributions are seen in Global Warming, Terrestrial Eutrophication (i.e. NO_x), Marine Eutrophication (i.e. NO_x and N), Ecotoxicity (i.e. As and Ni) and Depletion of Fossil Resources.

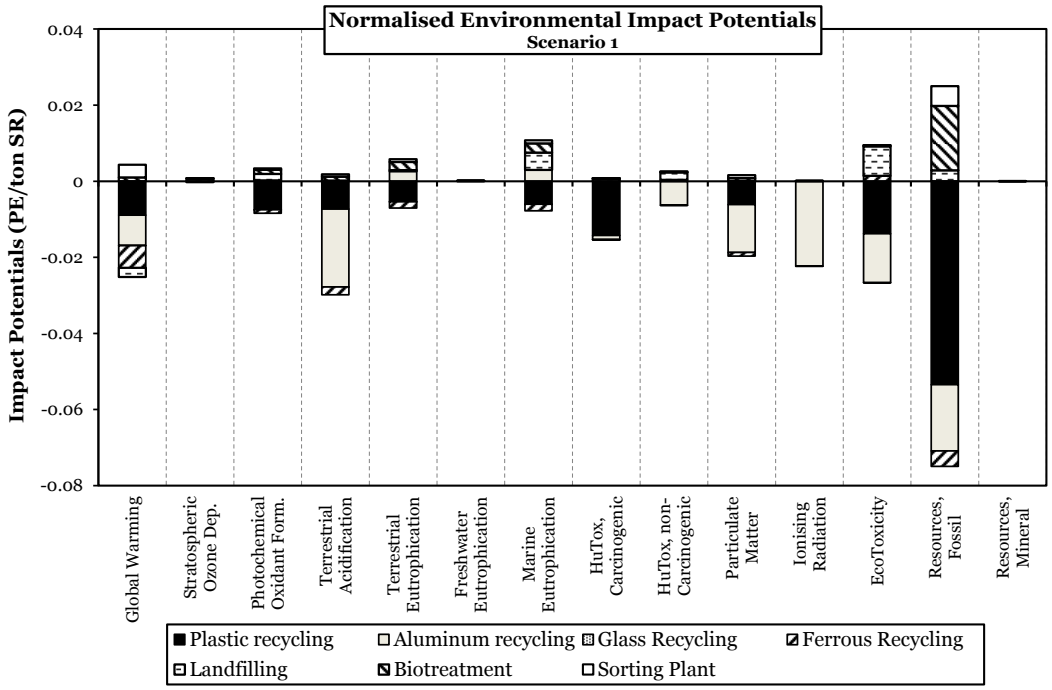


Figure 16 - Normalised potential impacts from the treatment of 1 ton of SR in Scenario 1, disaggregated according to individual treatment processes (PE = person equivalent).

5.2.2 Scenario 2

Disaggregated results for Scenario 2 are shown in Figure 17. In general, we see that the most significant contributions are associated with material recycling – plastic and aluminium – and the co-combustion of SR at a waste incinerator.

Estimated benefits are associated with avoiding emissions connected with the replacement of primary productions of materials and energy, namely NO_x (for Photochemical Oxidant formation), SO_x and some NO_x (for Acidification), NO_x (for Terrestrial Eutrophication) and NO_x (for Marine Eutrophication). Regarding human toxicity categories, savings are made from avoiding emissions of Cr^{VI} during plastic production (for Human Toxicity, Carcinogenic) and Hg from Al production (for Human Toxicity, non-Carcinogenic). Savings in Ecotoxicity and Particulate Matter are mainly related to the recycling of plastic and aluminium as well as SR co-incineration, and specifically to reduced emissions of V and Cr^{VI} (Ecotoxicity) and PM and SO_x (Particulate Matter).

With regards to burdens (i.e. positive impacts), significant contributions are seen in Human Toxicity non-Carcinogenic (i.e. Zn and some Hg) and Ecotoxicity (i.e. Zn), due to releases from the co-incineration process.

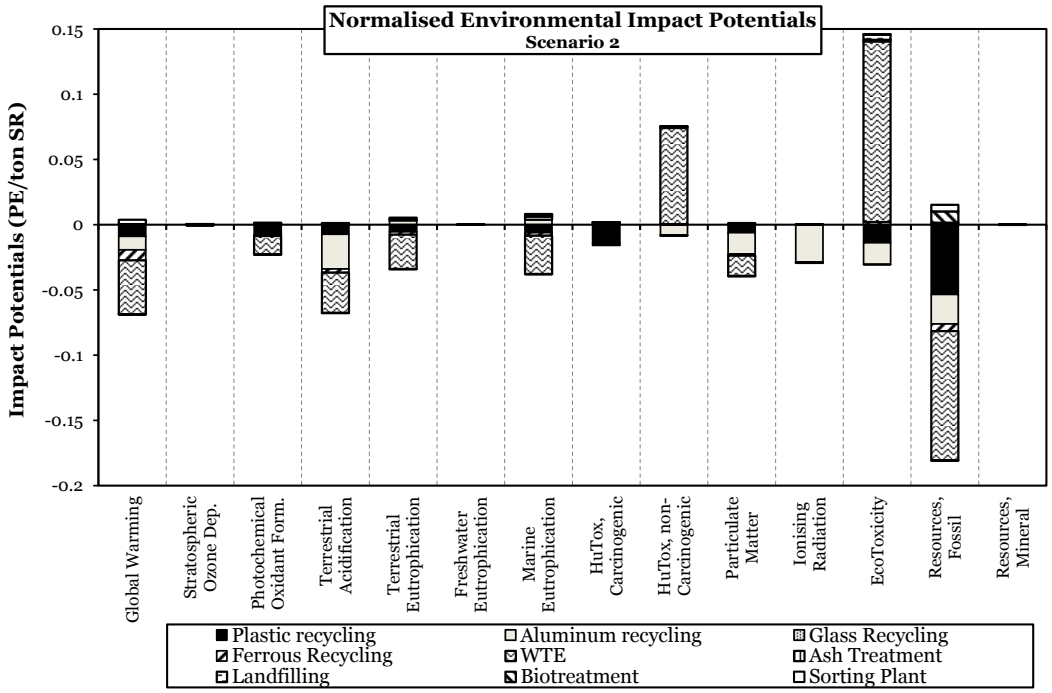


Figure 17 - Normalised potential toxic impacts from the treatment of 1 ton of SR in Scenario 2, disaggregated according to individual treatment processes (PE = person equivalent).

5.2.3 Scenario 3

Disaggregated results for Scenario 3 are shown in Figure 18. In general, we note that the most significant contributions are associated with the material recycling of plastic, aluminium and copper and with the pyrolysis of SR.

Estimated benefits are associated with avoiding emissions connected with the replacement of primary productions of materials and energy, namely NMVOC and NO_x (for Photochemical Oxidant formation), SO_x and some NO_x (for Acidification), NO_x (for Terrestrial Eutrophication), and NO_x (for Marine Eutrophication). Regarding human toxicity categories, savings are caused by avoiding emissions of Cr^{VI} during plastic production (for Human Toxicity, Carcinogenic) and Hg from aluminium production (for Human Toxicity, non-Carcinogenic). Savings in Ecotoxicity and Particulate Matter are mainly due to the recycling of plastic and aluminium, and specifically to reduced emissions of Cu and Cr^{VI} (Ecotoxicity) and PM (Particulate Matter).

With regards to burdens (i.e. positive impacts), significant contributions originate especially from the pyrolysis of >4 mm SR. The most significant emissions are NO_x (for Photochemical Oxidant formation), NO_x and SO_x (for Acidification), NO_x (for Terrestrial Eutrophication), P (for Freshwater Eutrophication), NO_x (for Marine Eutrophication), Zn (for Human Toxicity, non-Carcinogenic) and Cu and V (for Ecotoxicity).

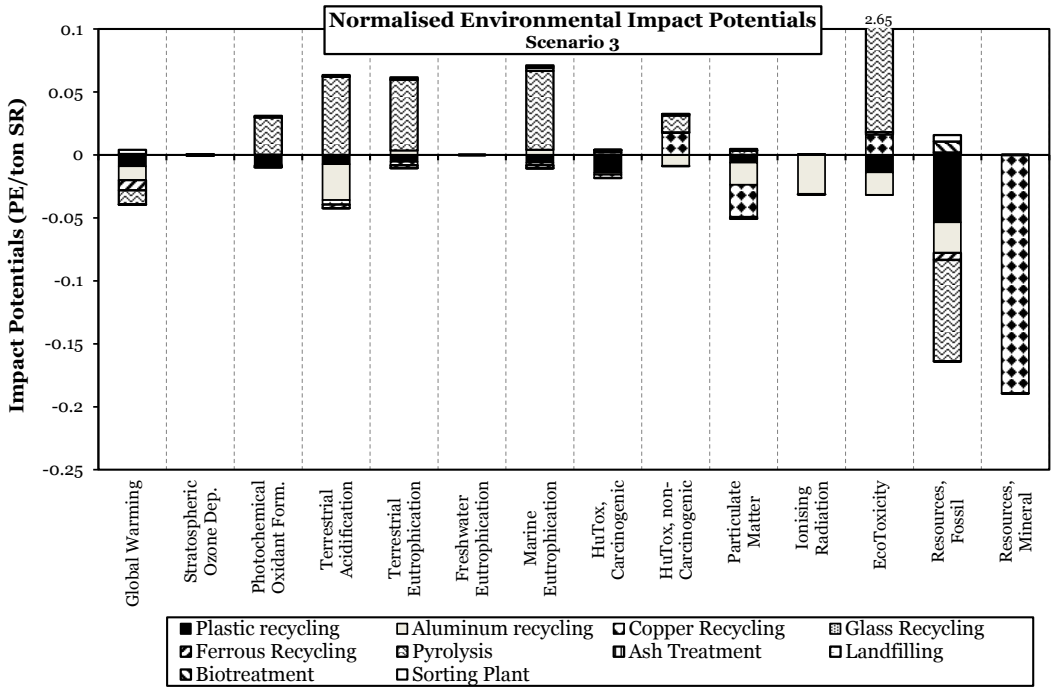


Figure 18 - Normalised potential toxic impacts from treatment of 1 ton SR in Scenario 3, disaggregated according to individual treatment processes (PE = person equivalent).

5.2.4 Scenario 4

Disaggregated results for Scenario 4 are shown in Figure 18. In general, it is evident that the most significant contributions are associated with material recycling – plastic, aluminium and copper – and the co-combustion of SR in a cement kiln.

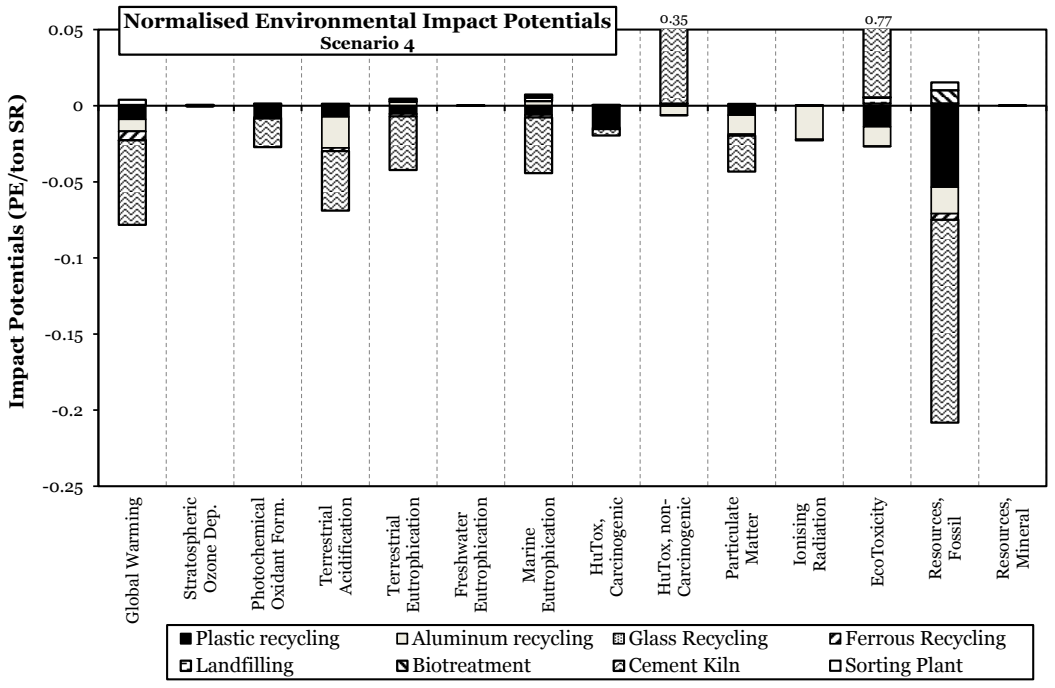


Figure 19 - Normalised potential toxic impacts from the treatment of 1 ton of SR in Scenario 4, disaggregated according to individual treatment processes (PE = person equivalent).

Estimated benefits are associated with avoiding emissions connected with the replacement of primary productions of materials and energy, namely NO_x and less NMVOC (for Photochemical

Oxidant formation), SO_x and NO_x (for Acidification), NO_x (for Terrestrial Eutrophication) and NO_x, and to a lesser contribution from ammonia (for Marine Eutrophication). Regarding human toxicity categories, savings are caused by avoiding emissions of Cr^{VI} and Hg during plastic production (for Human Toxicity, Carcinogenic) and Hg and As from aluminium production (for Human Toxicity, non-Carcinogenic). Savings in Ecotoxicity and Particulate Matter are related mainly to the recycling of plastic and aluminium, and specifically to reduced emissions of V and Cr^{VI} (Ecotoxicity) and PM and SO_x (Particulate Matter).

5.2.5 Conclusions regarding potential environmental impacts by process

The results are mostly influenced by the recycling of individual materials and thermal treatment, while SR landfilling generates limited potential impacts. The latter is related to the specific nature of SR, meaning that both the generation of gas and the release of metals into leachate are limited.

Results show that the recovery and recycling of plastic and metal fractions provide significant benefits and should thus be continued and possibly further improved. While for many impact categories emissions of SO_x and NO_x (either in foreground or background processes) may be the main contributors, significant improvements in scenarios 2, 3 and 4 could be achieved by limiting emissions into the air of Cu and Zn during thermal treatment and the related impacts on especially some of the toxic categories. This possibly could be achieved by properly pre-treating >4 mm residues, in order to reduce the content of metals in the material fed into such processes.

5.3 Potential environmental impacts by material fraction

In Figure 20 (and Figure 25 in Appendix 8), normalised potential impacts on Global Warming are presented as per unitary input (i.e. 1 ton) of individual material fractions. In general, it is noteworthy that the recycling of both ferrous and non-ferrous metals provides the greatest savings, and it should thus be the first priority when analysing and designing a management scheme for SR. The next material is plastic, which can provide significant savings, especially if different resins are effectively sorted and the least possible amount of cross-contamination is achieved.

With regards to the >4 mm residues fraction, the benefits potentially achievable are, to some extent, less significant than for the other fractions. The thermal treatment of >4 mm fractions induces significant improvements in the results for the impact on Global Warming, especially when looking at incineration and cement kilns. Results for pyrolysis of >4 mm residues are associated with very significant uncertainty, spanning from major savings in the case of optimal pyrolysis performance to positive burdens in the case of a bad configuration. As mentioned above, better data for pyrolysis are needed, in order to obtain more precise results, which can then be used as a basis for recommendations.

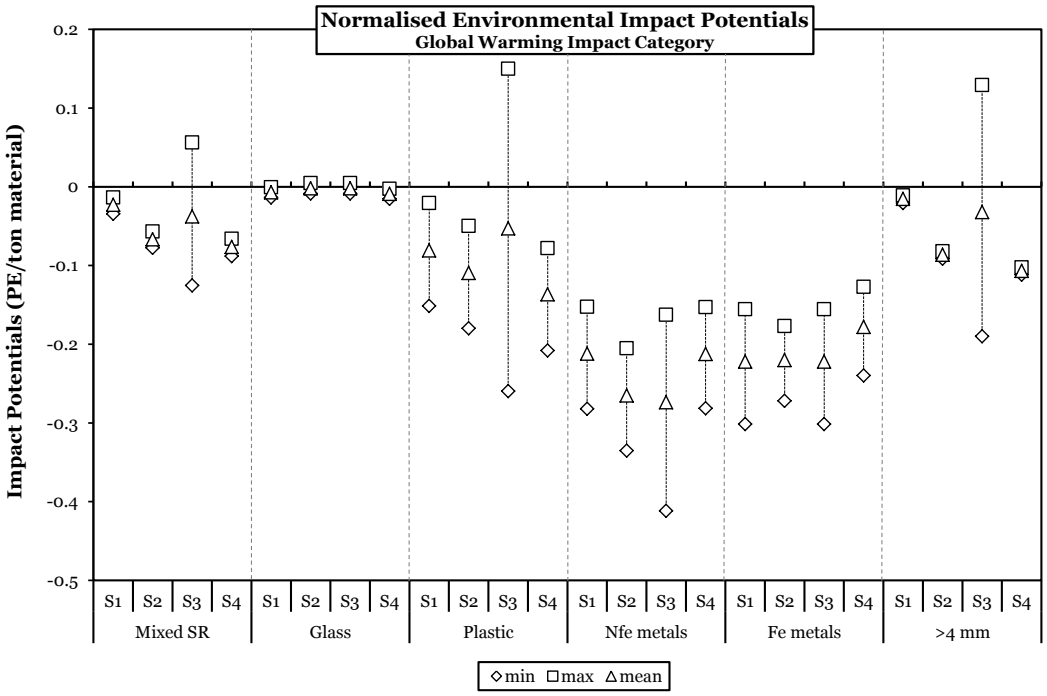


Figure 20 - Normalised potential Global Warming impacts in the four analysed scenarios, disaggregated according to individual material fractions and presented as per 1 ton of material (PE = person equivalent).

5.4 Sensitivity analysis

Sensitivity analysis is typically employed for identifying parameters, processes and datasets which make a significant contribution to the overall impacts of the analysed system and can thus be critical for results. As both contribution analysis and uncertainty quantification were performed in the previous sessions, the focus of the present sensitivity analysis is on the choice of background (i.e. marginal) electricity technology, which is often one of the most important decisions in an LCA, as both the usage and offsetting of electricity play a crucial role in overall savings. If the choice of marginal electricity was made as a part of the Monte Carlo simulation, it would be rather difficult to interpret whether the uncertainty was due to parameter variation or whether it was due to the scenario choice for the assumed electricity. It was therefore decided to perform a sensitivity analysis to assess what the impact would be if marginal electricity was not based on coal, as in the rest of the analysis, but instead on natural gas or wind power, which are possible future electricity sources in Denmark.

Datasets for electricity production relate to natural gas and coal. For this we used processes from the NEEDS (2008) project, which forecasted emissions associated with future energy generation. For electricity based on natural gas, a process for a 500MWe combined cycle plant was used. For wind power, a process for an offshore wind farm of 752 MW was used.

5.4.1 Sensitivity analysis of total potential environmental impacts

Potential impacts on non-toxic categories for the analysed scenarios in the sensitivity analysis are presented comparatively in Figure 21. The figure shows that for most impact categories the change in marginal electricity does not result in significant differences. Scenarios 1 (landfilling) and 4 (utilisation in a cement kiln) show increased savings, owing to the lower impacts resulting from the sorting and upgrading of materials, while no electricity is being offset. In this context, it should be mentioned that coal is still the energy source substituted in the cement kiln, as cement kilns will be operated on fossil fuels for several more decades to come. In the case of scenarios 2 (incineration) and 3 (pyrolysis), most impact categories show reduced overall savings, owing to the fact that the offset electricity has a lower impact.

In general, the ranking of scenarios according to their net impact is not affected by the change in marginal electricity. The only exception is the Global Warming category, demonstrating that when offsetting natural gas, incineration provides similar savings compared with landfilling, while the scenario based on pyrolysis is close to not offering a net saving. When considering wind power as the marginal electricity source, both scenarios based on incineration and pyrolysis have worse performance than the scenario based on landfilling. For all non-toxic impact categories, the scenario based on a cement kiln still seems the best option, as it is assumed that, even in the future, coal will still be used in kilns.

Potential impacts on toxic categories for the analysed scenarios in the sensitivity analysis are presented comparatively in Figure 22. The figure shows that for most impact categories, the results are not associated with significant changes, the reason being that most impacts on toxic categories are due to direct emissions or the offsetting of materials being recycled.

Potential impacts on Resource Depletion for the analysed scenarios in the sensitivity analysis are presented comparatively in Figure 23, which shows the same finding for fossil resources as for Global Warming potential, although in this case the ranking does not change. The reason for this is that even though there are very few resources offset from energy generation, fossil resources are still avoided through material recycling. For the depletion of mineral resources there is no change in impact when modifying marginal electricity.

The sensitivity analysis highlights that with the cleaner energies being offset, the more important it becomes to recycle plastic in SR, or sort it out in an RDF fraction for the cement kiln.

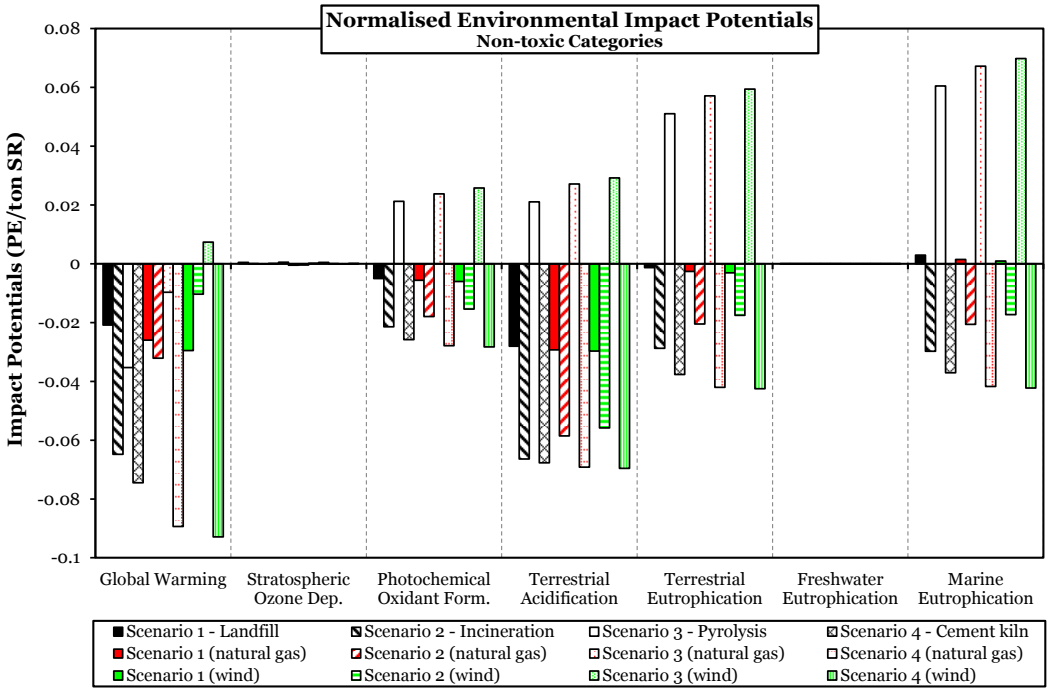


Figure 21 - Normalised potential non-toxic impacts from the treatment of 1 ton of SR in the four analysed scenarios modelled using alternative marginal electricity production (i.e. coal, natural gas and wind). (PE = person equivalent).

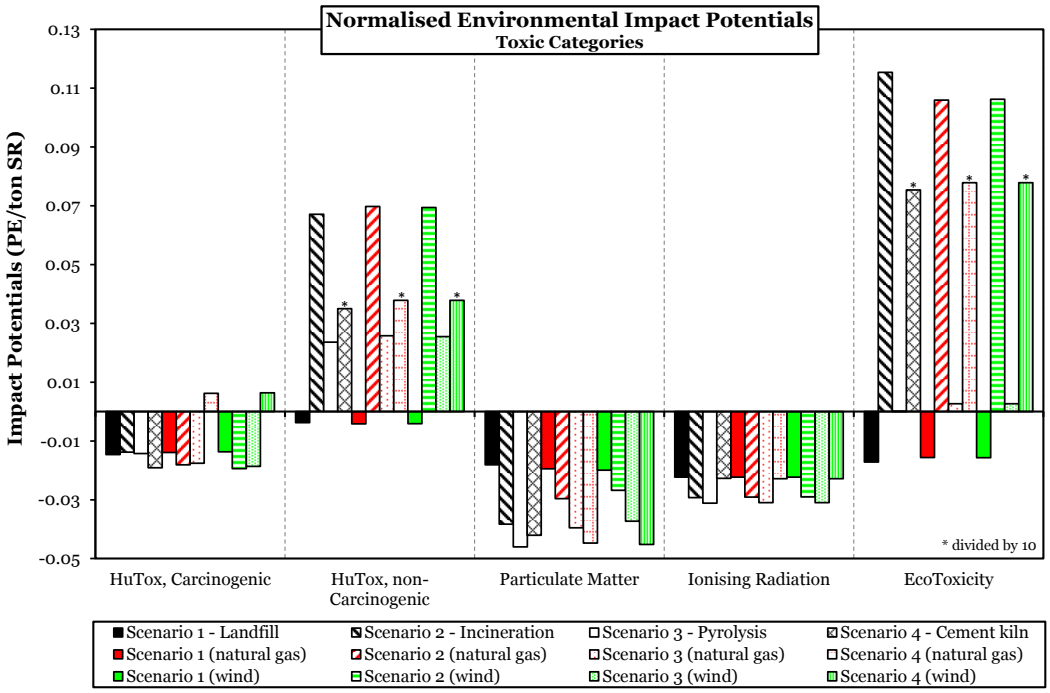


Figure 22 - Normalised potential toxic impacts from the treatment of 1 ton of SR in the four analysed scenarios modelled using alternative marginal electricity production (i.e. coal, natural gas and wind). (PE = person equivalent). Note that the following impacts were divided by 1,000: Ecotoxicity for Scenario 3 and 4; Human Toxicity non-Carcinogenic for Scenario 4.

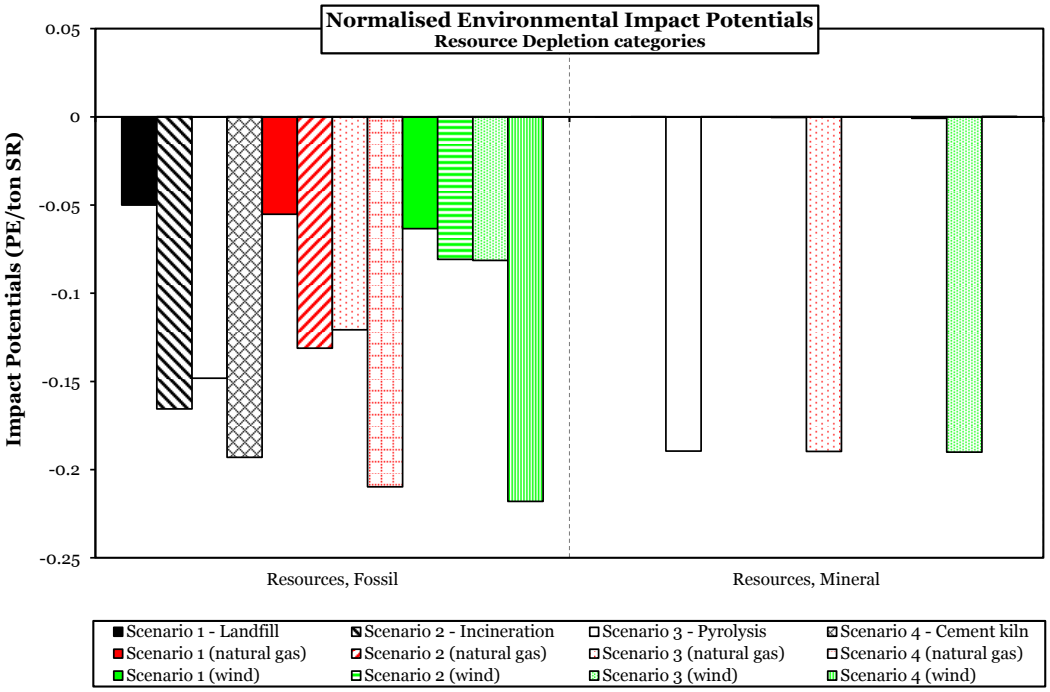


Figure 23 - Normalised potential Resource Depletion from the treatment of 1 ton of SR in the four analysed scenarios modelled using alternative marginal electricity production (i.e. coal, natural gas and wind). (PE = person equivalent).

6. Overall conclusions

Based on data and current knowledge:

- Diverting SR >4 mm fractions from landfill provides benefits from an environmental perspective and should thus be supported.
- Sorting and recycling metals, plastics and glass are beneficial to the environment and should thus be continued. Special focus should be given to increasing metal recovery, as this provides the greatest environmental benefits.
- Incineration seems currently the best option for the treatment of >4 mm residues.
- Pyrolysis seems to have worse energy efficiency but better downstream metal recovery (i.e. from the residues) than incineration. However, a clear conclusion could not be drawn, because results for pyrolysis were associated with significant uncertainty, owing to the lack of precise inventory data describing the process. Thus, any decision regarding the implementation of pyrolysis for treating >4 mm SR fractions should first be supported by pilot- to full-scale tests of the process, to deliver a better understanding thereof.
- The co-combustion of >4 mm fractions in a cement kiln could potentially provide significant savings for the Global Warming impact category. However, the current composition of the >4 mm fractions is not suitable, meaning that co-combustion in a cement kiln could only be implemented with a specially designed fraction. This option, in practice, would need a pre-treatment step to produce a feedstock with an RDF-like composition. This should especially include the additional screening and sorting of metals, to reduce the content of heavy metals in feedstock going into the kiln. Besides technical reasons, this would also result in a significant decrease in emissions into the air of heavy metals, which are largely responsible (especially Cu and Zn) for high potential impacts reported for some of the impact categories.
- Pre-treatment of the >4 mm SR should be considered, in order to reduce the content of heavy metals (especially Cu and Zn) in material fed into thermal processes.
- The results and recommendations are not significantly affected by the choice of the marginal technology, meaning they may still be valid within a different future energy system.

References

Ahmed, N., Wenzel, H., & Hansen, J. B. (2014). Characterization of Shredder Residues generated and deposited in Denmark. Waste Management, 34(7): 1279-88. doi: 10.1016/j.wasman.2014.03.017

Allegrini, E., Maresca, A., Emil, M., Sommer, M., Boldrin, A., & Astrup, T. F. (2014). Quantification of the Resource Recovery Potential of Municipal Solid Waste Incineration Bottom Ashes. Waste Management. doi: 10.1016/j.wasman.2014.05.003.

Astrup, T., Riber, C., & Pedersen, A. J. (2011a). Incinerator performance: effects of changes in waste input and furnace operation on air emissions and residues. Waste Management & Research, 29(10 Suppl), 57–68. doi:10.1177/0734242X11419893

Astrup, T., Dall, O., & Wenzel, H. (2011b). Fastlæggelse af energidata til brug i CO2-opgørelser. Part of the report: CO2 opgørelser i den danske affaldsbranche – en vejledning, affald danmark og Dakofa, København, oktober 2011.

Biganzoli, L., & Grosso, M. (2013). Aluminium recovery from waste incineration bottom ash, and its oxidation level. Waste Management & Research, 31(9), 954–9. doi:10.1177/0734242X13493956

Blok, K., Huijbregts, M., Roes, L., van Haaster, B., Patel, M., Hertwich, E., Wood, R., Hauschild, M.Z., Sellke, P., Antunes, P., Hellweg, S., Ciroth, A., Harmelink, M. (2013). Development and application of a standardized methodology for the PROspective SUstaInability assessment of TEchnologies (PROSUITE). Report prepared within the EC 7th Framework Project, n°: 227078.

Boldrin, A., Balzan, A., Astrup, T. (2013) Energy and environmental analysis of a rapeseed biorefinery conversion process. Biomass Conversion and Biorefinery, 3, 127–141. doi: 10.1007/s13399-013-0071-9

Boughton, B., & Horvath, A. (2006). Environmental assessment of shredder residue management. Resources, Conservation and Recycling, 47(1), 1–25. doi:10.1016/j.resconrec.2005.09.002

Cencic, O., & Rechberger, H. (2008). Material flow analysis with software STAN. Journal of Environmental Engineering & Management, 18(1), 3–7.

Christensen, T. H., Gentil, E., Boldrin, A., Larsen, A. W., Weidema, B. P., & Hauschild, M. (2009). C balance, carbon dioxide emissions and global warming potentials in LCA-modelling of waste management systems. Waste Management & Research, 27(8), 707–15. doi:10.1177/0734242X08096304

Ciacci, L., Morselli, L., Passarini, F., Santini, A., & Vassura, I. (2010). A comparison among different automotive shredder residue treatment processes. The International Journal of Life Cycle Assessment, 15(9), 896–906. doi:10.1007/s11367-010-0222-1

Classen, M., Althaus, H., & Blaser, S. (2007). Life cycle inventories of metals. Data v2.1. Final Report Ecoinvent Data. Ecoinvent v2.1 report No. 10, Ecoinvent Centre, Dübendorf, March 2009.

Clavreul, J., Baumeister, H., Christensen, T. H., and Damgaard, A. (2014). An environmental assessment system for environmental technologies. Environmental Modelling and Software, 60, 18-30. doi: 10.1016/j.envsoft.2014.06.007

Cramer, J., Malmgren-Hansen, B., Overgaard, J., & Larsen, O. H. (2006). Metoder til behandling af tungmetalholdigt affald - Fase 3. Miljøprojekt Nr. 1055, Miljøstyrelsen, Copenhagen, Denmark.

Daugaard, Anja Egede (2014). Personal communication with Anja Egede Daugaard, Senior Process Manager, FLSmidth A/S, Valby, Denmark.

De Marco, I., Caballero, B. M., Cabrero, M. a., Laresgoiti, M. F., Torres, a., & Chomón, M. J. (2007). Recycling of automobile shredder residues by means of pyrolysis. Journal of Analytical and Applied Pyrolysis, 79(1-2), 403–408. doi:10.1016/j.jaap.2006.12.002

Energistyrelsen. (2012). Technology Data for Energy Plants. Generation of electricity and district heating, energy storage and energy carrier generation and conversion. Energistyrelsen.

European Council (2000). Directive 2000/53/EC of the European Parliament and of the Council of 18 September 2000 on end-of life vehicles - Commission Statements. Official Journal L 269, 0034–0043, Brussels, 2000.

European Council (2003). December 2002 establishing criteria and procedures for the acceptance of waste at landfills pursuant to Article 16 of and Annex II to Directive 1999/31/EC. Official Journal of the European Communities. Brussels, 2002.

Fiore, S., Ruffino, B., & Zanetti, M. C. (2012). Automobile Shredder Residues in Italy: characterization and valorization opportunities. Waste Management, 32(8), 1548–59. doi:10.1016/j.wasman.2012.03.026

Fonseca, A. S., Nunes, M. I., Matos, M. A., & Gomes, A. P. (2013). Environmental impacts of end-of-life vehicles' management: recovery versus elimination. The International Journal of Life Cycle Assessment, 18(7), 1374–1385. doi:10.1007/s11367-013-0585-1

Forton, O. T., McGrady, L., Singh, M. M., Taylor, E. R. M., Moles, N. R., & Harder, M. K. (2007). Characterisation of rotary kiln residues from the pyrolysis of shredder residues: Issues with lead. Journal of Analytical and Applied Pyrolysis, 79(1-2), 395–402. doi:10.1016/j.jaap.2007.01.016

Franklin Associates. (2011). Life Cycle Inventory of 100% postconsumer HDPE and PET recycled resin from postconsumer containers and packaging. Prepared for the Plastics Division of The American Chemistry Council, Inc., the Association Of Postconsumer Plastic Recyclers (APR), the National Association For Pet Container Resources (NAPCOR), and the Pet Resin Association (PETRA).

Frischknecht, R., Jungbluth, N., Althaus, H., Doka, G., Dones, R., Heck, T., Hellweg, S., Hischier, R., Nemecek, T., Rebitzer, G., Spielmann, M., Wernet, G. (2007). Overview and methodology. Ecoinvent Rep. Ecoinvent v2.0 report No. 1, Ecoinvent Centre, Dübendorf, December 2007.

Galvagno, S., Fortuna, F., Cornacchia, G., Casu, S., Coppola, T., & Sharma, V. (2001). Pyrolysis process for treatment of automobile shredder residue: preliminary experimental results. Energy Conversion and Management, 42(5), 573–586. doi:10.1016/S0196-8904(00)00089-3

Hansen, J. B., Hjelmar, O., & Hyks, J. (2011a). Karakterisering af shredderaffald. Reno Djurs I/S, Rapport, February 2011.

Hansen, J. B., Hyks, J., & Hjelmar, O. (2011b). Deponering af shredderaffald Undersøgelse af driftsforhold til nedbringelse af efterbehandlingstiden. Reno Djurs I/S, Statusrapport 2010, Juli 2011.

Hansen, J. B., Hyks, J., Ahmed, N., Wenzel, H., Wellendorph, P., Nedenskov, J., Andersen, F., Thrane, J., Therkildsen, M. (2012). Lavteknologisk genanvendelse af ressource i deponeret shredderaffald via størrelsesfraktionering. Miljøprojekt Nr. 1440, Miljøstyrelsen, Copenhagen, Denmark.

Hischier, R. (2007). Life Cycle Inventories of Packagings & Graphical Papers. Data v2.0. Ecoinvent v2.0 report No. 11, Ecoinvent Centre, Dübendorf, December 2007.

Hyks, J., & Astrup, T. (2009). Influence of operational conditions, waste input and ageing on contaminant leaching from waste incinerator bottom ash: a full-scale study. Chemosphere, 76(9), 1178–84. doi:10.1016/j.chemosphere.2009.06.040

Hyks, J., Oberender, A., Hjelmar, O., Cimpan, C., Wenzel, H., Hu, G., & Cramer, J. (2014). Shredder residues: Problematic sub-stances in relation to resource recovery. Miljøprojekt Nr. 1568, Miljøstyrelsen, Copenhagen, Denmark.

Høstgaard, J., Andersen, K. R., Franck, N., Knudsen, N. O., Ahmed, N. S., Wenzel, H., Dall, O., Malmgren-Hansen, B., Nielsen, E.M., Olsen, R., Overgaard, J. (2012). Forbedret ressourceudnyttelse af shredderaffald. Miljøprojekt Nr. 1441, Miljøstyrelsen, Copenhagen, Denmark.

Jakobsen, J. B., Møller, J., Kromann, M., Neidel, T. L., & Jensen, M. B. (2013). Miljø-og samfundsøkonomisk vurdering af muligheder for øget genanvendelse af papir, pap, plast, metal og organisk affald fra dagrenovation. Miljøprojekt Nr. 1458, Miljøstyrelsen, Copenhagen, Denmark.

Joint Research Centre (JRC) (2011). European Commission Joint Research Centre - Institute for Environment and Sustainability: International Reference Life Cycle Data System (ILCD) Handbook- Recommendations for Life Cycle Impact Assessment in the European context. First edition November 2011. EUR 24571 EN. Luxemburg. Publications Office of the European Union; 2011

Malmgren-Hansen, B., Tønning, K., Cramer, J., Wismann, T., Overgaard, J., & Ottosen, L. M. (2002). Genanvendelseseffektivitet af hvidblik- og stålemballager. Miljøstyrelsen, Copenhagen, Denmark.

Mancini, G., Tamma, R., & Viotti, P. (2010). Thermal process of fluff: preliminary tests on a full-scale treatment plant. Waste Management, 30(8-9), 1670–82. doi:10.1016/j.wasman.2010.01.037

Mancini, G., Viotti, P., Luciano, A., & Fino, D. (2014). On the ASR and ASR thermal residues characterization of full scale treatment plant. Waste Management, 34(2):448-57. doi: 10.1016/j.wasman.2013.11.002.

Marco, I. De, Caballero, B., Torres, A., Laresgoiti, M. F., Chomón, M. J., & Cabrero, M. A. (2002). Recycling polymeric wastes by means of pyrolysis. Journal of Chemical Technology & Biotechnology, 77(7), 817–824. doi:10.1002/jctb.636

Mirabile, D., Pistelli, M. I., Marchesini, M., Falciani, R., & Chiappelli, L. (2002). Thermal valorisation of automobile shredder residue: injection in blast furnace. Waste Management, 22(8), 841–51.

Miljøstyrelsen (1998). Process from the EDIP database. Life Cycle Assessment of Packaging Systems for Beer and Soft Drinks, Environmental Project No 401, København, Denmark.

Morselli, L., Santini, A., Passarini, F., & Vassura, I. (2010). Automotive shredder residue (ASR) characterization for a valuable management. Waste Management, 30(11), 2228–34. doi:10.1016/j.wasman.2010.05.017

Møller, J., Damgaard, A., Brogaard, L. K.-S., & Astrup, T. F. (2014). Livscyklusvurdering af behandling af deponeret shredderaffald. Miljøprojekt Nr. [In press], Miljøstyrelsen, Copenhagen, Denmark.

Nedenskov, J. (2013). Forsøg med forbrænding af shredderaffald. Amagerforbrænding, Copenhagen, Denmark.

Nielsen, E. M., Hansen, H. J., Hundebøl, I., Hansen, K. W., Johansen, M. S., Nørholm, N. D., & Overgaard, J. (2006). Undersøgelse af mulighederne for termisk oparbejdning af mekanisk separeret shredderaffald. Miljøprojekt Nr. 1133, Miljøstyrelsen, Copenhagen, Denmark.

Nieminen, M., Suomalainen, M., & Mäkinen, T. (2006). Gasification of Shredder Residue. Research Notes 2344, VTT Tiedotteita, Espoo, Finland.

Nourreddine, M. (2007). Recycling of auto shredder residue. Journal of Hazardous Materials, 139(3), 481–90. doi:10.1016/j.jhazmat.2006.02.054

Osada, M., Tanigaki, N., Takahashi, S., & Sakai, S. (2008). Brominated flame retardants and heavy metals in automobile shredder residue (ASR) and their behavior in the melting process. Journal of Material Cycles and Waste Management, 10(2), 93–101. doi:10.1007/s10163-007-0204-y

Pasel, C., & Wanzl, W. (2003). Experimental investigations on reactor scale-up and optimisation of product quality in pyrolysis of shredder waste. Fuel Processing Technology, 80(1), 47–67. doi:10.1016/S0378-3820(02)00187-X

Passarini, F., Ciacci, L., Santini, A., Vassura, I., & Morselli, L. (2012). Auto shredder residue LCA: implications of ASR composition evolution. Journal of Cleaner Production, 23(1), 28–36. doi:10.1016/j.jclepro.2011.10.028

Poulsen, P. B., Westborg, S., & Cramer, J. (2011). Forprojekt til analyse af shredderaffald ifht. farlighed. Miljøprojekt Nr. 1374, Miljøstyrelsen, Copenhagen, Denmark.

Redin, L. a., Hjelt, M., & Marklund, S. (2001). Co-combustion of shredder residues and municipal solid waste in a Swedish municipal solid waste incinerator. Waste Management & Research, 19(6), 518–525. doi:10.1177/0734242X0101900607

Roy, C., & Chaala, a. (2001). Vacuum pyrolysis of automobile shredder residues. Resources, Conservation and Recycling, 32(1), 1–27. doi:10.1016/S0921-3449(00)00088-4

Ruffino, B., Fiore, S., & Zanetti, M. C. (2014). Strategies for the enhancement of automobile shredder residues (ASRs) recycling: results and cost assessment. Waste Management, 34(1), 148–55. doi:10.1016/j.wasman.2013.09.025

Santini, A., Morselli, L., Passarini, F., Vassura, I., Di Carlo, S., & Bonino, F. (2011). End-of-Life Vehicles management: Italian material and energy recovery efficiency. Waste Management, 31(3), 489–94. doi:10.1016/j.wasman.2010.09.015

Saxena, S., Rao, N., Rehmat, A., & Mensinger, M. (1995). Combustion and co-combustion of auto fluff. Energy, 20(9), 877–887. doi: 10.1016/0360-5442(95)00038-I

Scheutz, C., Fredenslund, A. M., Nedenskov, J., & Kjeldsen, P. (2010a). Release and fate of fluorocarbons in a shredder residue landfill cell: 1. Laboratory experiments. Waste Management, 30(11), 2153–62. doi:10.1016/j.wasman.2010.03.035

Scheutz, C., Fredenslund, A. M., Nedenskov, J., & Kjeldsen, P. (2010b). Release and fate of fluorocarbons in a shredder residue landfill cell: 2. Field investigations. Waste Management, 30(11), 2163–9. doi:10.1016/j.wasman.2010.03.033

Scheutz, C., Fredenslund, A. M., Nedenskov, J., Samuelsson, J., & Kjeldsen, P. (2011). Gas production, composition and emission at a modern disposal site receiving waste with a low-organic content. Waste Management, 31(5), 946–55. doi:10.1016/j.wasman.2010.10.021

Seerup, U. (2012). Tillæg til miljøgodkendelse. For: I/S Reno-Nord. Godkendelsen omfatter: Forbrænding af shredderaffald. Miljøstyrelsen, J.nr. MST-1270-00687, Århus, Denmark.

Shen, Z., Day, M., Cooney, J. D., Lu, G., Briens, C. L., & Bergougnou, M. a. (1995). Ultrapyrolysis of automobile shredder residue. The Canadian Journal of Chemical Engineering, 73(3), 357–366. doi:10.1002/cjce.5450730313

Skatteministeriet (2011). Lovbekendtgørelse nr. 311 af 1. april 2011 om afgift af affald og råstoffer (Consolidation Act on fee on waste and raw materials). Skatteministeriet, Copenhagen, Denmak.

Vermeulen, I., Block, C., Van Caneghem, J., Dewulf, W., Sikdar, S. K., & Vandecasteele, C. (2012). Sustainability assessment of industrial waste treatment processes: The case of automotive shredder residue. Resources, Conservation and Recycling, 69, 17–28. doi:10.1016/j.resconrec.2012.08.010

Weidema, B., & Wesnæs, M. (1996). Data quality management for life cycle inventories—an example of using data quality indicators. Journal of Cleaner Production, 4(3), 167–174. doi: 10.1016/S0959-6526(96)00043-1

Zolezzi, M., Nicolella, C., Ferrara, S., Iacobucci, C., & Rovatti, M. (2004). Conventional and fast pyrolysis of automobile shredder residues (ASR). Waste Management, 24(7), 691–9. doi:10.1016/j.wasman.2003.12.005

Appendix 1: List of processes from databases used in the LCA

Table 15 shows the datasets retrieved from LCA databases and used in EASETECH to model the treatment of SR.

Table 15 – Overview of datasets used for LCA modelling of SR treatment.

Database	Process
EASETECH	Road, Long haul truck, Euro3, 25t, Generic, 2006
EASETECH	Marginal Electricity Consumption incl. Fuel Production, Coal, Energy Quality, DK, kWh, 2006
EASETECH	Electricity, natural gas, at power plant, NORDEL
EASETECH	District Heating, marginal average, (DK), kWh, 2012
EASETECH	Natural Gas in Industry Burner (prod + comb), >100 kW, 1996
EASETECH	Extraction of sub-base material - gravel (L/S 10)
EASETECH	Aluminium, Al (Primary), World average, 2005
EASETECH	LPG (prod + comb), kg, TERMINATED, 1990
EASETECH	LIMITED - Glass - bottle (primary), EDIP, 1990
EASETECH	Natural gas, EU-27, ELCD, 2002 - corrected
EASETECH	Water from Waterworks, Sweden, 2008
EASETECH	Collection Vehicle, 10t Euro3, urban traffic, 1 liter diesel, 2006
EASETECH	Hard Coal in Power Plant (prod + comb), EU-15, 2003
EASETECH	Oxygen, EU-27, ELCD, 2005 - corrected
EASETECH	Steel Sheets (97.75% primary), Sweden, 2008
EASETECH	District Heating, Scania county, kWh, SE, 2008
EASETECH	Polyethylene high density granulate (PE-HD), RER, ELCD, 1999 - corrected
EASETECH	brass, at plant, CH
EASETECH	Process water, RER, ELCD, 2005 - corrected
EASETECH	LIMITED - Hydrated Lime, CaOH2, EU-27, ELCD, 2007
EASETECH	lead, primary, at plant, GLO
Ecoinvent 2.2	Light fuel oil, burned in industrial furnace 1 M, non-modulating, RER, modified
EASETECH (EDIP, IPU-NF-E2752)	Production and Combustion of Diesel Oil in Truck, EU2, 1998
EASETECH (UUID)	Sodium chloride (NaCl), RER, ELCD, 1996 - corrected
EASETECH (UUID)	Sodium hydroxide (NaOH), RER, ELCD, 1996 - corrected
Ecoinvent 2.2	Polyethylene, HDPE, granulate, at plant, RER
Ecoinvent 2.2	Copper, primary, at refinery, RER
Ecoinvent 2.2	Copper, secondary, at refinery, RER
Ecoinvent 2.2	Lead, secondary, at plant, RER
Ecoinvent 2.2	Zinc, primary, at regional storage, RER
Ecoinvent 2.2	Heat, natural gas, at boiler atmospheric low-NOx non-modulating <100kW, RER
ELCD	LIMITED - Limestone, CaCO3, EU-27, ELCD, 2006
LIPASTO ILCD	Wheel loader, combustion 1L of diesel, 2003/20
LIPASTO ILCD	Earth moving lorry, Gross vehicle mass 32t, pay load capacity 19t,Highway driving,EURO4 LIPASTO
LIPASTO ILCD	Bulldozer, combustion 1L of diesel, 2003/2011

Appendix 2: Chemical composition of newly produced shredder residues

An overview of literature studies providing composition data for SR is provided in Table 16, where data are divided according to geographical origin. Table 16 shows that there are considerable differences in the type of shredder residues included in the studies. While in other countries most data represent ASR, the majority of data for Denmark focus on SR as a mixture of ASR and waste from other sources, indicating that data may not be directly comparable.

Table 16 – Literature overview of composition data on shredder residues.

Geographical area	Reference	Dataset description
Denmark	Cramer et al. (2006)	SR-1996
		SR-2002
		SR-2002
	Nielsen et al. (2006)	ASR (100%)
		SR (80 SR-20% ASR)
	Nieminen et al. (2006)	ASR (100%)
		SR
		SR-2005
	Hansen et al. (2011a)	SR
	Nedenskov (2013)	SR- Prøve 1
		SR-Prøve 2
	Poulsen et al. (2011)	SR-Del 1
		SR-Del 2
		SR-Del 3
	Høstgaard (2012)	SR-1 (with <4mm)
		SR-2 (without <4mm)
Italy	Galvagno et al. (2001)	ASR-min
		ASR-max
	Mirabile et al. (2002)	ASR
	Zolezzi et al. (2004)	ASR
	Mancini et al. (2010)	ASR
		Fluff 1
		Fluff 2
		Fluff 3
	Morselli et al. (2010)	ASR
	Santini et al. (2011)	ASR-min
		ASR-max
	Ruffino et al. (2014)	ASR
	Fiore (2012)	SR
		ASR1
		ASR2
		LF
		LF < 4 mm
		LF > 4 mm,
	Mancini et al. (2014)	ASR-2007
		ASR-2008
Japan	Osada et al. (2008)	ASR
Korea	Roh (2013)	ASR
Spain	de Marco (2007)	Light ASR
		Heavy ASR
Sweden	Nourredine (2007)	ASR
United States	Saxena et al. (1995)	ASR

Table 17 presents physicochemical characteristics of typical Danish SR and ASR. For individual parameters, a ratio (based on average values) between SR and ASR is calculated to enable an overall comparison of the two types of materials. The results indicate that compared with ASR, SR present a higher content of metals (e.g. Al, Fe, Mn) and a lower content of combustibles (e.g. C, LHV). SR is also enriched with some of the critical heavy metals (e.g. Cr, Cu, Mo, Ni, Pb, Zn), possibly because of the industrial origin of a large share of these residues.

Table 17 – Physicochemical characteristics of Danish SR based on literature data (SR= shredder residues; ASR= automotive shredder residues).

		SR, DK (20% ASR, 80% SR)			ASR, DK (100% ASR)			SR/ASR
	Unit	Min	Max	Aver.	Min	Max	Aver.	
Dry matter	%	87.1	91.7	89.4	96.9	96.9	96.9	0.92
Ash	% TS	33.4	69.7	46.6	43.1	47.5	45.3	1.03
C	% TS	16.12	50.2	31.1	42.2	42.4	42.3	0.74
N	% TS	0.3	1.3	0.8	1.4	1.8	1.6	0.50
LHV	MJ/kg	9.4	21.3	14.5	14.9	14.9	14.9	0.98
Si	mg/kg TS	400	91000	47213.6	7200	7200	7200	6.56
Al	mg/kg TS	126	26000	16779.4	12000	12000	12000	1.40
Ca	mg/kg TS	305	35000	19058.6	5000	5000	5000	3.81
Fe	mg/kg TS	830	132000	61400.8	6500	6500	6500	9.45
K	mg/kg TS	27	3700	1921.0	280	280	280	6.86
Mg	mg/kg TS	60	6500	3920.0	6300	6300	6300	0.62
Mn	mg/kg TS	0	1200	548.6	90	90	90	6.10
Na	mg/kg TS	61	9600	5352.9	12000	12000	12000	0.45
P	mg/kg TS	11	820	392.4	80	80	80	4.91
S	mg/kg TS	60	2800	1829.0	2700	3700	3200	0.57
Ti	mg/kg TS	43	3500	2050.6	490	490	490	4.18
As	mg/kg TS	1	36	18.9	0	0		
Ag	mg/kg TS	0	0		0	0		
Ba	mg/kg TS	29	2800	1259.9	1590	1590	1590	0.79
Be	mg/kg TS	0.53	0.53	0.5	0	0		
Cd	mg/kg TS	15	40	27.1	0	0		
Co	mg/kg TS	28	51	38.0	0	0		
Cr	mg/kg TS	4	890	306.8	70	70	70	4.38
Cu	mg/kg TS	19	25000	14019.1	660	660	660	21.24
Hg	mg/kg TS	0.82	1.6	1.2	0.8	0.8	0.8	1.51
Mo	mg/kg TS	10	53	23.6	2	2	2	11.79
Nb	mg/kg TS	-	-	-	0	0		
Ni	mg/kg TS	2	480	245.8	20	20	20	12.29
Pb	mg/kg TS	14	13000	3668.0	140	140	140	26.20
Si	mg/kg TS	2300	2300	2300.0	0	0		
Sb	mg/kg TS	3	300	159.0	0	0		
Sc	mg/kg TS	-	-	-	0	0		
Se	mg/kg TS	1.4	1.4	1.4	0	0		
Sn	mg/kg TS	2	180	73.4	10	10	10	7.34
Sr	mg/kg TS	2	370	160.6	30	30	30	5.35
V	mg/kg TS	39	61	53.8	0	0		
W	mg/kg TS	130	130	130.0	0	0		
Y	mg/kg TS	7	7	7.0	0	0		
Zn	mg/kg TS	243	20000	12503.8	1000	1000	1000	12.50
Zr	mg/kg TS	2	270	76.5	20	20	20	3.83
Br	mg/kg TS	10	2100	532.5	0	0		
Cl	mg/kg TS	84	26000	15153.3	17000	24000	20500	0.74
F	mg/kg TS	220	350	303.3	100	270	185	1.64
I	mg/kg TS	4.9	4.9	4.9	0	0		0.92
		Cramer et al. (2006); Nielsen et al. (2006); Nieminen et al. (2006); Hansen et al. (2011a); Nedenskov (2013); Poulsen et al. (2011); Høstgaard et al. (2012)			Nielsen et al. (2006); Nieminen et al. (2006)			

As international literature data could not be used and data on ASR seemed not fully appropriate (Table 17), it was decided to use data provided by the few available Danish studies dealing with SR. While Table 16 and Table 17 reports composition of mixed SR, physicochemical composition of sub-fractions of SR was also needed to assess management alternatives of individual fractions (see scenario description in Chapter 3). Only Hansen et al. (2011a) and Høstgaard et al. (2012) reported both grain size distribution, material fraction and chemical compositions of Danish SR. The results from these two studies are summarized in Table 18, Table 19 and Table 20.

Table 18 - Grain size distribution of Danish SR.

	Grain size	average (%)	st.dev.
Hansen et al. (2011a)	> 10 mm	37.11	9.69
	10 - 4 mm	17.35	1.18
	4 - 1 mm	14.31	2.92
	< 1 mm	31.23	7.21
	Total	100	
Høstgaard et al. (2012)	> 6 mm	64.5	16.1
	3-6 mm	6.4	0.5
	< 3 mm	29.0	16.6
	Total	100	

Table 19 – Material fraction composition of Danish SR with particle size >10 mm (based on Hansen et al., 2011a).

	> 10 mm average (%)	st.dev.
Metals	14.5	2.6
Plastic	35.6	2.0
Rubber	20.6	4.4
Wood	7.1	0.3
Foam/Fluff	6.3	0.3
Textile	0.8	0.1
Paper/Cardboard	0.3	0.3
Wires	3.1	0.1
Electronics	0.4	0.0
Glass/Ceramics	0.1	0.0
Stones	5.1	0.0
Misc /Mix	6.1	0.2
Total	100	

Table 20 – Material fraction composition of Danish SR with particle sizes 3-6 mm and >6 mm (based on Høstgaard et al., 2012).

Material fraction	>6 mm average (%)	st.dev	3-6 mm average (%)	st.dev
Ferrous metal	1.25	0.67	9.21	0.05
Non-Ferrous metal	4.32	0.22	4.27	0.13
Tin can	0.90	0.10	0.30	0.12
Plastic	29.13	1.52	8.82	0.90
Rubber	9.89	0.07	1.68	0.36
Wood	9.62	0.17	8.14	1.24
Foam	4.06	0.85	1.39	0.64
Electronic	2.97	1.45	1.48	0.24
Dirt/inert	6.34	1.80	6.88	0.48
Other	31.52	0.21	57.84	0.59
Total	100		100	

For this project, it was considered that recovery of recyclables (i.e. glass, plastic and metals, see scenario description in Chapter 3) was performed on the >10 mm fraction, and thus the material fraction composition reported in Table 19 was used for the LCI modelling. This choice was also based on the fact that the composition reported in Table 20 includes a large amount of undefined

materials (i.e. 'Other') adding additional uncertainty to the modelling. However, as Hansen et al. (2011a) did only provide content of 'Metals' without further specify the type of metals, the relative distribution between 'Ferrous metal', 'Non-Ferrous metal' and 'Tin can' provided by Høstgaard et al. (2012) for >6mm fraction (Table 21) was used to estimate the content of individual metals in the waste composition. According to Table 20, the distribution of metals is shown in Table 22:

Table 21 – Relative distribution of metal fractions in Høstgaard et al. (2012).

Fraction	Data in Table 20 (%)		Relative distribution (%)	
	average	stdev	average	stdev
Ferrous metal	1.25	0.67	19.31	10.51
Non-Ferrous metal	4.32	0.22	66.74	8.08
Tin can	0.90	0.10	13.95	2.17
Total	6.48	0.71		

The residues after recovery of glass, plastic and metals would then be separated by means of a screen into two fractions, namely >4 mm and <4 mm. Combining data from Table 18 and Table 19, and assuming that recovery of glass, plastic and metals (in Table 19) reaches 90%, the material fraction composition reported in Table 22 is used for LCI modelling of Danish SR. Within uncertainty ranges, the data reported in Table 22 are very much in line with results from two sampling/sorting campaigns conducted by Stena Recycling in April 2014 at their Roskilde facility (these data cannot be disclosed because of confidentiality reasons). The error reported in Table 22 was estimated based on uncertainty values reported in Table 18 and Table 19 and applying common rules for error propagation.

Table 22 – Material fraction composition used for LCI modelling of SR.

Fraction	average (%)	st.dev.
Ferrous metal	0.93	0.53
Non-Ferrous metal	3.22	0.99
Tin can	0.67	0.21
Plastic	11.88	2.86
Rubber	6.89	2.10
Glass	0.03	0.01
>4mm	30.83	3.72
<4mm	45.54	3.15
Total	100	

The average physicochemical composition of fractions <4mm and >4mm (shown in Table 23 and Table 24) was estimated based on data provided by Hansen et al. (2011a) and Høstgaard et al. (2012). The geometrical mean and deviation were estimated according to the following formulas:

Geometric mean e^{μ}

Deviation $(e^{\sigma^2} - 1)e^{2\mu+\sigma^2}$

where μ and σ are, respectively, the mean and standard deviation of the log-transformed data. As Hansen et al. (2011a) and Høstgaard et al. (2012) did not include data regarding water content in SR, it was decided to use values previously reported by Nieminen et al. (2006) for Danish SR: 89.4 ± 3.25% on wet weight.

Table 23 - Physicochemical composition of <4mm fractions of SR used in the LCI modelling.

		Hansen et al. (2011a)		Høstgaard et al. (2012)				
		4-1 mm	<1 mm	Prøv 1	Prøv 2	Prøv 3	geom mean	deviation
Ash	% TS	72.20	82.10	83.69	83.69	83.69	80.9	21.6
LHV	MJ/kgTS	8.05	3.60	0.05	0.05	0.05	0.33	4.12
C	% TS	17.20	9.60	2.75	2.75	2.75	5.09	9.26
H	% TS	2.26	1.56	3.24	3.24	3.24	2.60	1.90
N	% TS	0.54	0.42	0.50	0.50	0.50	0.49	0.16
Si	mg/kg TS	93500	146000	115276	69740	107325	103328	66307
Al	mg/kg TS	39706	33708	21127	24854	27552	28668	17252
Ca	mg/kg TS	35467	34210	48749	84918	62339	50020	41523
Fe	mg/kg TS	188000	188409	151399	237249	207149	192386	88205
K	mg/kg TS	4104	6001	5610	9528	3301	5341	4609
Mg	mg/kg TS	6828	7202	10213	11016	12020	9216	5670
Mn	mg/kg TS	1914	1884	1601	2707	2406	2066	1107
Na	mg/kg TS	13244	12403	13327	8010	12119	11627	6289
P	mg/kg TS	836	1282	2204	5317	496	1442	2778
Ti	mg/kg TS	3573	5091	8112	4488	7408	5472	4163
As	mg/kg TS	52	33	401			88.4	286.1
Ba	mg/kg TS	3648	2991	2396	3805	3207	3168	1559
Be	mg/kg TS	1	1				0.64	0.19
Cd	mg/kg TS	14	44				25.2	41.0
Co	mg/kg TS	30	57				41.1	38.8
Cr	mg/kg TS	4110	1010	801	902	300	979	1947
Cu	mg/kg TS	44500	1950	1900	2800	1700	3793	13183
Hg	mg/kg TS	1	3				1.51	2.33
Mo	mg/kg TS	167	59				99.0	149.2
Nb	mg/kg TS	4	5				4.47	2.00
Ni	mg/kg TS	722	658	300	601	802	585	490
Pb	mg/kg TS	7000	2810	1499	2700	2907	2971	3356
S	mg/kg TS	2070	2740	7602	9719	8718	5159	7568
Sb	mg/kg TS	259	134	2207	500	200.0	377	923
Sc	mg/kg TS	1	1				0.95	0.27
Sn	mg/kg TS	951	338	200	501	401	419	488
Sr	mg/kg TS	765	616	200	300	401	408	453
V	mg/kg TS	97	82				89.1	34.6
W	mg/kg TS	166	216				189	95
Y	mg/kg TS	7	12				9.47	7.63
Zn	mg/kg TS	17400	25200	29696	71376	76336	37170	49679
Zr	mg/kg TS	377	472	601	601	802	553	366
Br	mg/kg TS	360	60	7927	500	401	510	2672
Cl	mg/kg TS	7240	2450	2565	7288	5101	4422	4887
I	mg/kg TS	2	4				2.78	2.60
Li (*)		0.190	0.013				0.05	0.31

Table 24 - Physicochemical composition of >4mm fractions of SR used in the LCI modelling.

		Hansen et al. (2011a)	Høstgaard et al. (2012)				
		> 4 mm	Prøv 1	Prøv 2	Prøv 3	geom mean	deviation
Ash	% TS	55.00	36.40	36.40	36.40	40.4	21.4
LHV	MJ/kgTS	14.36	18.09	18.09	18.09	17.1	6.3
C	% TS	33.50	36.00	36.00	36.00	35.4	6.9
H	% TS	4.32	5.19	5.19	5.19	4.96	1.61
N	% TS	0.92	0.80	0.80	0.80	0.83	0.23
Si	mg/kg TS	79000	46300	34800	51000	50477	38061
Al	mg/kg TS	31500	14500	11300	14600	16568	15484
Ca	mg/kg TS	39100	36400	30400	27700	33087	14866
Fe	mg/kg TS	55500	101700	74900	69800	73703	44498
K	mg/kg TS	3260	3200	2600	3000	3003	1039
Mg	mg/kg TS	7570	6900	7100	6900	7112	1536
Mn	mg/kg TS	1120	1300	900	900	1042	506
Na	mg/kg TS	8070	6700	5600	7300	6857	3033
P	mg/kg TS	700	1300	1100	1400	1088	767
Ti	mg/kg TS	2840	5100	7500	5300	4898	4219
As	mg/kg TS	24	100			49.2	106.8
Ba	mg/kg TS	3460	3300	2600	1400	2539	2246
Be	mg/kg TS	1				0.62	
Cd	mg/kg TS	10				9.73	
Co	mg/kg TS	36				35.8	
Cr	mg/kg TS	517	500	300	300	391	272
Cu	mg/kg TS	42100	2600	7700	3500	7370	21697
Hg	mg/kg TS	3				2.61	
Mo	mg/kg TS	40				39.6	
Nb	mg/kg TS						
Ni	mg/kg TS	979	300	300	200	364	510
Pb	mg/kg TS	8610	1800	2700	1100	2605	4788
S	mg/kg TS	1680	7000	4900	4200	3944	4889
Sb	mg/kg TS	212	400	500	300	336	271
Sc	mg/kg TS	1				1.20	
Sn	mg/kg TS	464	200	200	100	208	266
Sr	mg/kg TS	296	200	300	100	205	218
V	mg/kg TS	39				39.1	
W	mg/kg TS	179				179	
Y	mg/kg TS	4				4.49	
Zn	mg/kg TS	24000	30600	27400	17300	24290	14551
Zr	mg/kg TS	198	300	300	200	244	142
Br	mg/kg TS	558	1300	500	100	436	1033
Cl	mg/kg TS	19300	11300	10300	4800	10190	11939
I	mg/kg TS	2				1.88	
Li (*)							

Appendix 3: LCI of combustion of SR at a waste incinerator

Jakobsen et al. (2013) provided a comprehensive inventory dataset for a state-of-the-art waste incineration plant in Denmark (Table 26). This inventory is based on data collected from state-of-the-art Line 5 of Vestforbrænding I/S in 2012. The flue gas cleaning system includes wet scrubbing for acid gases removal, SNCR deNOx system and activated carbon filter for dioxin and Hg removal. The dataset is based on both process and input specific emissions, depending on whether the emissions are mostly controlled by the plant operation or are mainly related to the content of specific compounds in the waste input.

The inventory from Jakobsen et al. (2013) is hereby used to model the combustion of SR at a waste incinerator in Denmark. The choice is explained in details in the following sections, where definition of uncertainty related to individual parameters is also estimated based on data from Vermeulen et al. (2012), Nedenskov (2013) and Mancini et al. (2014).

Flue gas

Process-specific and input-specific emissions reported in Jakobsen et al. (2013) are considered valid and thus used upon the condition that ASR is co-combusted with regular municipal solid waste up to a share of 12-14% and based on the following considerations:

- Astrup et al. (2011a) tested co-combustion by blending 14% ASR into municipal solid waste. Table 25 presents composition of waste used in the test, the stack emission of specific compounds and the respective TCs. It is seen that despite the content of individual compounds significantly increased with the introduction of ASR in the feedstock blend (compared with municipal residual waste), emissions from the stack were not affected. In some cases, TCs to air seemed eventually to decrease with the addition of ASR, possibly because a significant share of the metals is likely to be embedded in a non-combustible matrix and are thus not volatilized to the flue gas phase. This may suggest that TCs for ASR are somehow lower than for residual waste and it is hence conservatively assumed that TCs for ASR equal to regular waste. While results are only available for As, Cd, Cr, Pb, and SB, a similar behaviour for other compounds is also assumed.

Table 25 – TCs to air for 100% municipal residual waste and 14% blend of ASR, according to Astrup et al. (2011a).

	100% normal waste (NW)			NW (86%) + ASR (14%)		
	Stack (g)	Waste (g)	TC (%)	Stack (g)	Waste (g)	TC (%)
As	0.0029	10	0.029	0.0008	12.3	0.0065
Cd	0.0066	6	0.11	0.0025	8.5	0.0294
Cr	0.0035	105	0.0033	0.0053	182	0.0029
Pb	0.125	290	0.0431	0.081	945	0.0085
Sb	0.022	34	0.0647	0.01	83	0.012

- Both Redin et al. (2001), Astrup et al. (2011a), Nedenskov (2011) and Vermeulen et al. (2012) reported regular functioning of the flue gas cleaning system and rather constant emissions during co-combustion of ASR with municipal solid waste. Redin et al. (2001) tested 20% ASR, Astrup et al. (2011a) performed a co-combustion test with 14% ASR on a mass basis, while Nedenskov (2011) tested a mix with 12.4% ASR and Vermeulen et al. (2012) a mix with 25 and 39% ASR.
- The Danish EPA deliberated to I/S Reno-Nord an environmental approval for co-combustion of ASR with municipal solid waste up to 12.5% on mass basis (Seerup, 2012). In the approval, the Danish EPA assessed that the flue gas cleaning system at I / S Reno-Nord will ensure that the emission of heavy metals to air emissions do not rise, or at least not rise significantly, as a consequence of ASR co-combustion.

Table 26 – Inventory dataset for combustion of SR at a waste incinerator.

Type	Parameter	Unit	Jakobsen et al. (2013)	Deviation
Input material	NaOH	kg/ton ww	0.024	
	Activated carbon	kg/ton ww	1.04	
	CaCO3	kg/ton ww	5.67	
	NH3	kg/ton ww	1.53	
	Water	kg/ton ww	397	
	Ca(OH)2	kg/ton ww	0.34	
	Polymer	kg/ton ww	0.0006	
	HCl	kg/ton ww	0.0056	
	TMT	kg/ton ww	0.395	
Output	Electricity	% LHV, net	22	
	Heat	% LHV, net	73	
Process-specific air emission	HCl	kg/ton ww	0.0053	
	CO	kg/ton ww	0.033	
	NOx	kg/ton ww	0.849	
	HF	kg/ton ww	0.00039	
	Dioxin	kg/ton ww	$1.8*10^{-11}$	
	PM	kg/ton ww	0.003	
	SO2	kg/ton ww	0.00291	
Input-specific air emissions (% of content in waste input)	Cl	% input	0.1073	0.3036
	S	% input	0.099	0.2801
	As	% input	0.0121	0.0366
	Cd	% input	0.0064	0.0174
	Cr	% input	0.0394	0.1336
	Cu	% input	0.00261	0.0092
	Hg	% input	0.7476	2.0014
	Ni	% input	0.0329	0.1007
	Pb	% input	0.00081	0.0023
	Sb	% input	0.0119	0.0337
	Zn*	% input	0.0717	0.0994
Bottom ash composition (% of content in waste input)	Cl	% input	5.3	2.81
	S	% input	23.99	12.70
	As	% input	40.62	21.50
	Cd	% input	11.83	6.26
	Cr	% input	83.15	44.01
	Cu	% input	92.63	49.03
	Fe	% input	96.92	51.30
	Hg	% input	2.38	1.26
	Mo	% input	96.61	51.14
	Ni	% input	87.32	46.22
	Pb	% input	48.47	25.65
	Sb	% input	38.91	20.59
	Se	% input	22.38	11.85
	Zn	% input	51.76	27.40
Fly ash composition (% of content in waste input)	Cl	% input	32.13	13.69
	S	% input	60.91	25.95
	As	% input	58.92	25.10
	Cd	% input	88.13	37.55
	Cr	% input	16.77	7.15
	Cu	% input	7.35	3.13
	Fe	% input	3.06	1.30
	Hg	% input	96.25	41.01
	Mo	% input	2.54	1.08
	Ni	% input	12.56	5.35
	Pb	% input	51.29	21.85
	Sb	% input	59.84	25.50
	Se	% input	76.73	32.69
	Zn	% input	48.18	20.53
* calculated based on Vermeulen et al. (2012)				

Vermeulen et al. (2012) investigated incineration of 100% ASR in a rotary kiln plant, reporting process-specific emissions very similar to Jakobsen et al. (2013) (Table 26). With regards to input-

specific emissions, Vermeulen et al. (2012) reported slightly higher TCs for some of the assessed compounds (Table 28). The small differences (e.g. Cd and Cr) may be associated to the fact that, while Jakobsen et al. (2013) covered a grate furnace incineration plant, Vermeulen et al. (2012) performed a test on a rotary kiln process used for incineration of hazardous waste, thus running on higher temperature.

Based on the abovementioned, data from Vermeulen et al. (2012) were used to estimate relative uncertainty in flue gas emissions, as reported in Table 26. Vermeulen et al. (2012) provided data (reported in Table 27) for both the waste composition used in the test and the amount of individual compounds emitted with the flue gas. In addition, data from Vermeulen et al. (2012) were also used to estimate TC to air for Zn, as it was missing in Jakobsen et al. (2013).

The TCs to air (TC_{air}) were calculated according as:

$$TC_{air} = \frac{Output_{air}}{Input_{waste}}$$

where $Input_{waste}$ is the content of a specific compound in the waste and $Output_{air}$ is the emission of such compound to the atmosphere. Using a uniform distribution for $Output_{air}$ (minimum and maximum showed in Table 27) and a lognormal distribution for $Input_{waste}$ (assuming average as the geometrical mean and mix/max as the 2.5/97.5% range Table 27), the TC_{air} where computed 15000 times. The obtained results were characterized by a log-normal distribution, for which median and variance were calculated, as shown in Table 28. For individual compounds, a coefficient of variation (CV) was calculated (see Table 28) and applied to TCs reported by Jakobsen et al. (2013) to derive the data variance reported in Table 26. The CVs were calculated as follows:

$$\text{CV (\%)} = \frac{\text{variance}}{\text{median}}$$

The number of simulations was decided based on the following formula:

$$\varepsilon = \frac{3\sigma}{\sqrt{N}} \rightarrow N = \left(\frac{3 \times \sigma}{\varepsilon}\right)^2 = \left(\frac{3 \times stdevp(min, max, average(min, max))}{50}\right)^2$$

where ε is the error of the estimation (here assumed maximum 2%), σ is the deviation of the data and N the number of simulations. The value N was estimated for all individual compounds using data in Table 27, and a rounding of the maximum value (i.e. 14999) being then selected for running the simulation.

Table 27 – Data regarding emissions to air and waste composition, as provided by Vermeulen et al. (2012).

Compound	Unit	Air emission		Waste composition		
		min	max	min	max	average
C	kg/ton$_{SR}$	1.503		279	626	410
N	kg/ton$_{SR}$			8.8	45	19
S	kg/ton$_{SR}$	0.0044	0.0148	1.9	5.6	3.7
As	g/ton$_{SR}$	3.03E-07	1.90E-02	1.2	70	30
Cd	g/ton$_{SR}$	3.08E-03	3.07E-01	2	86	34
Cr	g/ton$_{SR}$	7.09E-08	5.74E-03	17	7000	1120
Cu	g/ton$_{SR}$	3.93E-06	3.44E+00	27	16,600	4910
Hg	g/ton$_{SR}$	9.08E-03	1.66E-01	0.2	14	4.1
Ni	g/ton$_{SR}$	1.91E-06	5.87E-01	54	4000	734
Pb	g/ton$_{SR}$	1.04E-01	8.64E+00	94	7000	2610
Zn	g/ton$_{SR}$	5.29E+00	5.78E+00	1430	14,100	8260

Table 28 – TC_{air} for combustion of SR incineration plant calculated based on data from Vermeulen et al. (2012).

Compound	Median	Variance	CV (%)

As	0.026	0.079	302
Cd	0.354	0.961	272
Cr	0.000	0.000	339
Cu	0.021	0.073	351
Hg	1.233	3.301	268
Ni	0.014	0.044	306
Pb	0.124	0.357	287
Zn	0.072	0.099	139
S	0.255	0.241	94.8

Solid residues

Table 29 presents that ratio between the amount of individual compounds ending in bottom ash (BA) and fly ash (FA). This can also be interpreted as a ratio between the TCs to BA and FA. It can be seen that the distribution of different substances to solid outputs varies significantly among individual incineration plants. It is thus not possible to combine the data to univocally define an average dataset to be used for estimating to uncertainty in solid distribution.

It was thus decided to estimate the uncertainty of TCs using data from Mancini et al. (2014), who present repeated measurement on the same test system. Composition data on input waste and solid outputs were used to reconcile the mass and material balance of the system using STAN (Cencic and Rechberger, 2008). Data reconciliation allowed calculating the TCs - to bottom ash, fly ash and flue gas – and their respective uncertainty, as shown in Table 30. For individual compounds, CVs were calculated as follows:

$$\text{CV (\%)} = \frac{\text{variance}}{\text{median}}$$

Using individual CVs, an average CV for TC to BA (i.e. 53%) and FA (i.e. 43%) were estimated (Table 30). These two CVs were then used to derive uncertainty (i.e. deviation) for TCs to BA and FA reported in Table 26 by multiplying individual TCs with the CVs.

Table 29 – Ratio BA:FA according to Jakobsen et al. (2013), Astrup et al. (2011a) and Mancini et al. (2014). The ration is calculated for individual compounds by dividing the amount ending in bottom ash (BA) and fly ash (FA). For Mancini et al. (2014), FA includes both boiler ash and APC residues.

	Ratio BA:FA		
	Jakobsen et al. (2013)	**Astrup et al. (2011a)**	**Mancini et al. (2014)**
As	0.69	2.33	0.71
Cd	0.13	0.19	0.02
Cl	0.16	0.25	
Cr	4.96	8.09	2.70
Cu	12.60	24.00	3.40
Fe	31.67		
Hg	0.02	0.02	3.37
Mo	38.04	3.55	
Ni	6.95	32.33	2.57
Pb	0.95	3.55	0.48
S	0.39	0.45	
Sb	0.65	1.44	
Se	0.29		0.49
Zn	1.07	1.63	0.18

Table 30 – TCs to fly ash and bottom ash, and respective CVs for incineration of ASR according to Mancini et al. (2014).

	Fly ash			**Bottom ash**		
	TC (%)	**st.dev.**	**CV(%)**	**TC (%)**	**st.dev.**	**CV(%)**
As	0.392	0.111	28.42	0.280	0.169	60.34

Ash	0.158	0.057	35.79	0.410	0.178	43.41
Cd	0.955	0.544	56.94	0.018	0.023	131.32
Cr	0.069	0.034	49.56	0.185	0.072	39.01
Cu	0.194	0.136	69.96	0.661	0.689	104.29
TS	0.063	0.011	17.89	0.169	0.010	5.89
Hg	0.024	0.007	26.75	0.082	0.032	38.55
Ni	0.120	0.056	46.72	0.309	0.102	33.14
Pb	0.471	0.202	42.80	0.225	0.107	47.48
Se	1.439	0.407	28.30	0.700	0.347	49.60
VS	0.010	0.003	32.57	0.037	0.010	25.98
Zn	1.362	1.030	75.61	0.240	0.135	56.15
Average			42.61			52.93

Leaching from solid residues

Solid residues from combustion of SR in incineration plant are assumed to be used as sub-road basis material during road construction (
Figure 3 and
Figure 4). Owing to water infiltration, leaching of different compounds from the solid residues may occur with subsequent potential adverse impacts on the environment.

A univocal estimation of leaching coefficients from SR solid residues is however not feasible because SR would usually be incinerated in co-combustion with other waste materials (Seerup, 2012), typically a mixture of municipal and industrial solid waste. The effects of co-combustion on the chemistry of solid residues is rather complicated, meaning that estimating leaching factors specific for SR allocated according to the share of the co-fuels is not possible.

Table 31 provides a comparison of leaching factors with increasing share of SR in the feedstock to the incineration process. It is seen that individual compounds show different behaviours, meaning that general conclusions cannot be drawn. This is in line with Hyks et al. (2014), that reported no clear correlation between the leaching of metals from residues and the increasing total content of metals in the residues when increasing the share of (A)SR in the feedstock. The release of some compounds (e.g. Ba, Cl, S) from solid residues seems to increase with increasing amount of SR in the feedstock, while for others (i.e. Cr, Cu, Mn, Ni, Zn) it seems to decrease, finally a number of compounds (i.e. As, Cd, Hg, Mo, Pb, Sb, Se, F, Na) do not show a specific behaviour.

Results in Table 31 show that a consistent pattern across all compounds could not be determined and that using leaching factors measured for 100% ASR material (e.g. from Mancini et al., 2014) may not be correct. It was hence decided to use in the LCI modelling the leaching data originated from a feedstock with 12.4% SR (from Nedenskov, 2013), as this was considered the most realistic process condition. The interval ranges from Nedenskov (2013) reported in Table 31 were entered in the LCI modelling as uniform distributions.

Table 31 – Comparison of leaching from solid residues originating from incineration of SR, MSW and co-combustion.

		100% MSW Nedenskov (2013)	12.4% SR Nedenskov (2013)	100% ASR Mancini et al. (2014)	Tendency
As	µg/l	1.4-7.4	4.9-6.1	0.25-0.37	↔
Ba	µg/l	0.35-110	133-145	700-1133	↑
Cd	µg/l	0.04-0.2	0.133-0.150	0.06-0.1	↔
Cr	µg/l	8.1-380	15.0-19.0	0.62-1.43	↓
Cu	µg/l	195-1.900	315-550	<0.01	↓
Hg	µg/l	0.05-0.32	<0.2	0.08-1	↔
Mn	µg/l	150-1.000	<0.2-0.370		↓
Mo	µg/l			26.15-46.6	↔
Ni	µg/l	1-13	4.9-5.0	0.02-0.04	↓
Pb	µg/l	0.5-32	0.4-3.0	5.57-10.4	↔
Sb	µg/l			58.2-88.01	↔
Se	µg/l		2.9-3.7	1.41-2.05	↔
Zn	µg/l	5-90	8.7-13.0	<0.01	↓
Cl	mg/l	0.830-2.0	1.2-1.3	118.8-143.4	↑
F	mg/l			0.81-0.96	↔
S	mg/l	0.15-2.8	0.9-0.9	8.21-9.51	↑
Na	mg/l	0.10-1.50	0.8-0.963		↔
DOC	mg/l			<5	↔
↑: increasing with increasing ASR share; ↓: decreasing with increasing ASR share; ↔: no clear tendency					

Appendix 4: LCI of pyrolysis of SR

The LCI of pyrolysis of SR was created as an average dataset based on available literature. Data regarding mass (Table 32) and energy (Table 33) distribution as well as chemical composition of char (Table 34 and Table 35) and oil (Table 36) products from the pyrolysis process were used to calculate average value and their uncertainty. These mean values (and their uncertainty) were then used to reconcile the mass, energy and substance balances of the SR pyrolysis system using STAN (Cencic and Rechberger, 2008), as shown in Figure 24, Figure 25, Figure 26, Figure 27, Figure 28, Figure 29. Data reconciliation allowed calculating the TCs to individual outputs - to char, oil, gas – and their respective uncertainty. An overview of the LCI dataset of pyrolysis of SR is finally shown in Table 37.

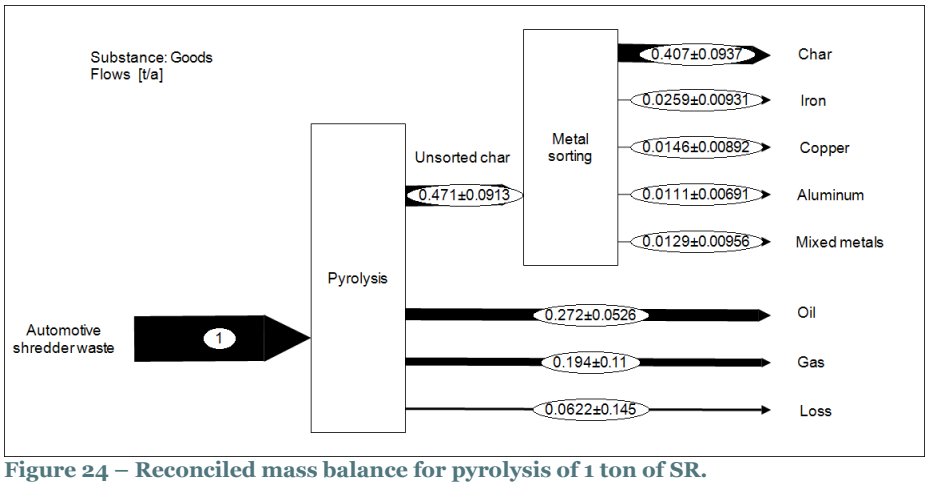


Figure 24 – Reconciled mass balance for pyrolysis of 1 ton of SR.

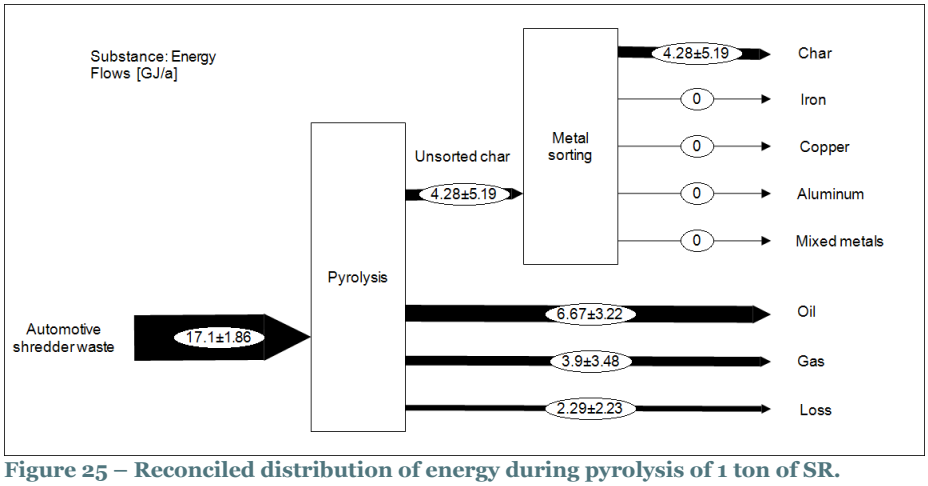


Figure 25 – Reconciled distribution of energy during pyrolysis of 1 ton of SR.

Table 32 - Literature overview of mass transfer during pyrolysis of ASR.

		Input	Output						
Source	**Test**	**Waste**	**Char**	**Oil**	**Gas**	**Loss**	**Metals**	**Slag**	**Fly ash**
		ton	**% input**	**% input**	**% input**	**% input**	**kg/ton$_{ASR}$**	**kg/ton$_{ASR}$**	**kg/ton$_{ASR}$**
/A/	700 °C	1	68	4	14	14			
	750 °C	1	64	7	17	12			
/B/	680 °C	1	43.57	20.07	13	23.36			
	600 °C	1	44.55	32.62	9.04	13.79			
	550 °C	1	59.28	19.52	4.23	16.97			
/C/		1	35	40	20	5			
/D/	500 °C	1	55	25.7	5.8	13.5			
	600 °C	1	52.3	30	11.5	6.2			
	700 °C	1	39	24.2	24.1	12.7			
	800 °C	1	37	21.8	34	7.2			
	500 °C	1	47.5	35	7.7	9.8			
	600 °C	1	37.5	43	5.8	13.7			
	700 °C	1	33	55	8.5	3.5			
	800 °C	1	25	58	12.2	4.8			
/E/	400 °C	1	69.4	7.1	23.5				
	500 °C	1	63.6	10.3	26.1				
	700 °C	1	58.8	8.8	32.4				
	400 °C	1	54.1	25.5	20.4				
	500 °C	1	39.4	29	31.6				
	700 °C	1	38.3	20.2	41.5				
/F/	400 °C	1	38.7	12.3	49				
	500 °C	1	42.2	13.6	44.2				
	600 °C	1	45.8	18.2	36				
	700 °C	1	47.4	13.4	39.2				
	800 °C	1	44.6	14.4	41				
/G/	F1	1	75.49	16.01	3.8	4.7			
	F2	1	69.09	22.35	3.73	4.83			
/H/		1	65	17	9	9			
/I/		1					58	170	39
/J/		1					53		
mean		1	49.7	23.0	21.0	10.3	55.5	170	39
st.dev.		0	13.2	13.6	14.2	5.39	3.54		

/A/: Shen et al. (1995); /B/: Galvagno et al. (2001); /C/: Pasel and Wanzl (2003) /D/: Zolezzi et al. (2004); /E/: de Marco et al. (2007); /F/: Joung et al. (2007); /G/: Høstgaard et al. (2012); /H/: Santini et al. (2012); /I/: Roh et al. (2013); /J/: Møller et al. (2014)

Table 33 – Literature overview of energy transfer during pyrolysis of ASR.

		Input				Output			
Source	**Test**	**Waste**	**Electricity**	**Pyrolysis gas**	**Pyrolysis oil**	**Char**	**Oil**	**Gas**	**Loss**
		GJ	**MJ**	**GJ**	**GJ**	**% input**	**% input**	**% input**	**% input**
/A/	680	17.4				22.1	34.7	15.9	27.3
	600	17.4				22.6	56.3	8.25	12.8
	550	17.4				30.1	33.7	3.28	32.9
/B/	500	21				23.9	42.0	1.70	32.5
	600	21				20.4	49.0	8.42	22.2
	700	21				8.13	39.5	16.4	36.0
	800	21				8.42	35.6	28.4	27.6
	500	21				41.1	60.5	0.14	-1.74
	600	21				25.6	72.5	0.47	1.47
	700	21				22.2	78.0	2.36	-2.64
	800	21				16.5	79.6	4.33	-0.37
/C/	400					43.7	22.8	27.2	6.38
	500					26.0	33.0	37.8	3.14
	700					26.1	26.9	45.7	1.27
	400					42.3	36.6	11.5	9.66
	500					22.8	40.9	24.6	11.7
	700					22.6	28.1	29.6	19.7
/D/	400					27.8	23.4	48.8	
	500					21.5	26.0	52.5	
	600					17.1	33.9	49.0	
	700					16.1	24.8	59.1	
	800					17.1	26.5	56.4	
/E/	F1	10.8				61.0	33.2	2.40	3.49
	F2	15.0				53.9	33.8	1.65	10.6
/F/		17.6	248.4	0.29	0.21	65.3	1.65	33.0	
mean		18.8	248.4	0.29	0.21	28.2	38.9	22.7	13.4
st.dev.		3.11				14.9	18.3	20.1	12.9

/A/: Galvagno et al. (2001); /B/: Zolezzi et al. (2004); /C/: de Marco et al. (2007); /D/: Joung et al. (2007); /E/: Høstgaard et al. (2012); /F/: Møller et al. (2014)

Table 34 – Typical physicochemical composition of char products from pyrolysis of ASR and SR (Part 1).

Char		Ash	VS	H	C	N	O	S	Cl	P	K	Al	Na	Mg	Si	Ba
		% ww	% ww	% ww	% ww	% ww	% ww	% ww	% ww	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg
/A/	680			0.94	21.5	0.55		1.46	0.88							
	600			1.37	19.9	0.54		1.38	0.54							
	550			3.86	41.5	1.09		1.11	0.96							
/B/		68.0	32.0	0.60	28.6	0.69			2.30			14100				
/C/	500			1.30	28.0	0.00	13.5	0.51								
	600			0.70	27.0	0.00	12.1	0.84								
	700			0.30	16.9	1.00	10.3	0.00								
	800			0.39	15.6	0.00	6.73	0.73								
	500			4.00	43.7	0.10	14.5	0.36								
	600			1.70	39.5	0.00	10.0	1.00								
	700			0.86	40.0	0.00	5.00	0.68								
	800			0.69	40.0	0.00	5.70	1.26								
/D/	400	73.7		1.60	20.8	0.90		0.60								
	500	82.0		0.80	16.2	0.80		0.70								
	700	87.4		0.30	16.8	0.70		0.70								
	400	33.2		4.00	60.2	0.90		0.70								
	500	44.7		1.50	50.4	1.10		0.90								
	700	46.0		0.80	53.4	1.00		0.90								
/E/	SR															
	ASR															
/F/	400											4291				
	500											6127				
	600											8304				
	700											9509				
	800											12636				
/G/	Dp 13			2.99	30.8	0.81		0.95	5.30	1200	4300	14700	3900	10500	66000	6300
mean		62.1	32.0	1.51	32.1	0.54	9.73	0.82	2.00	1200	4300	9952	3900	10500	66000	6300
st.dev.		20.8		1.25	13.8	0.44	3.60	0.36	1.97			4011				

/A/: Galvagno et al. (2001); /B/: Roy and Chaala (2001); /C/: Zolezzi et al. (2004); /D/: de Marco et al. (2007); /E/: Forton et al. (2007) /F/: Joung et al. (2007); /G/: Høstgaard et al. (2012)

Table 35 – Typical physicochemical composition of char products from pyrolysis of ASR and SR (Part 2).

Char		Mn	Ca	Cd	Co	Cr	Cu	Fe	Hg	Ni	Pb	Sb	V	Zn	Ti
		mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg
/A/	680														
	600														
	550														
/B/		1471	50200	73.0	118	223	15000	121000		1179	4300		43.0	14100	
/C/	500														
	600														
	700														
	800														
	500														
	600														
	700														
	800														
/D/	400														
	500														
	700														
	400														
	500														
	700														
/E/	SR						34624	44393			6220			14108	
	ASR						18809	42856			2446			8930	
/F/	400			10.2		135	15450	71604	0.08	129	2028		12.7	7336	
	500			13.1		195	22198	39214	0.03	162	3069		14.1	6222	
	600			19.7		143	14339	47824	0.02	217	2250		15.6	8561	
	700			8.50		123	8203	65025	0.02	206	2110		16.5	7622	
	800			4.80		90.3	8983	89432	0.08	205	1533		19.0	5235	
/G/	Dp 13	1700	80000			900	1700	123000			3500	700		35000	9600
mean		1586	65100	21.6	118	258	15479	71594	0.0	350	3051	700	20.2	11901	9600
st.dev.		162	21072			286	9416	32833	0.0	408	1460		11.4	9213	

/A/: Galvagno et al. (2001); /B/: Roy and Chaala (2001); /C/: Zolezzi et al. (2004); /D/: de Marco et al. (2007); /E/: Forton et al. (2007) /F/: Joung et al. (2007); /G/: Høstgaard et al. (2012)

Table 36 – Typical physicochemical composition of oil products from pyrolysis of ASR and SR.

	Oil	H	C	N	O	S	Cl	Al	Cd	Co	Cr	Cu	Fe	Hg	Mn	Ni	Pb	V	Zn
		% ww	% ww	% ww	% ww	% ww	% ww	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg
/A/	1	10.9	87.3	0.98	0.48	0.34		5.39		2.70	4.65	1.67	413			2.70	2.70		10.7
	2	10.4	78.8	2.46	8.07	0.27		8.74		10.5	2.23	3.53	1202			4.37	22.7		52.5
	3	11.4	81.3	2.63	4.42	0.25		6.42		11.5	1.58	3.07	290			3.26	10.3		22.5
	4	10.8	84.3	1.5	2.8	0.5	0.1			14.0	7.44			0.372			9.3		67.0
	5	10.7	84.2	0.69	3.56	0.72	0.13	0.53		0.22		0.26	51.6		0.31	0.12			9.11
/B/	Heavy ASR	9.6	86.2	1		0.3													
	Light ASR	9.6	81.1	1.5		0.4													
/C/	600	8	74.3	0	15.2	0													
	500	11	66	0.92	12.5	0.15													
	600	11.2	64.4	0	15.6	0.3													
	700	8.9	59.3	1.7	19	0.27													
	800	8.5	58	1.5	19	0.7													
/D/	500	9.6	81.1	1.5		0.4													
	700	8.9	82.9	1.6		0.2													
	400	10.4	85.2	1.1		0.3													
	500	9.4	86.1	1		0.4													
	700	8.8	86.8	1.9		0.4													
/E/	400							5.16	0.004		2.67	0.26	27.2			2.88	1.97	0.005	20.5
	500							7.20	0.014		1.71	1.021	61.3			6.67	5.25	0.10	22.7
	600								0.02		1.09		8.62			0.64	6.42	0.013	
	700							7.48			0.38		6.92				0.98	0.032	32.1
	800								0.03		0.23	2.795	7.22				2.5		
mean		9.89	78.1	1.29	10.06	0.35	0.12	5.85	0.02	7.78	2.44	1.80	230	0.37	0.31	2.95	6.90	0.04	29.6
stdev		1.04	9.92	0.71	7.03	0.18	0.02	2.65	0.01	5.97	2.30	1.35	393			2.21	6.77	0.04	20.3

/A/: Roy and Chaala (2001); /B/: de Marco et al. (2002); /C/: Zolezzi et al. (2004); /D/: de Marco et al. (2007); /E/: Joung et al. (2007)

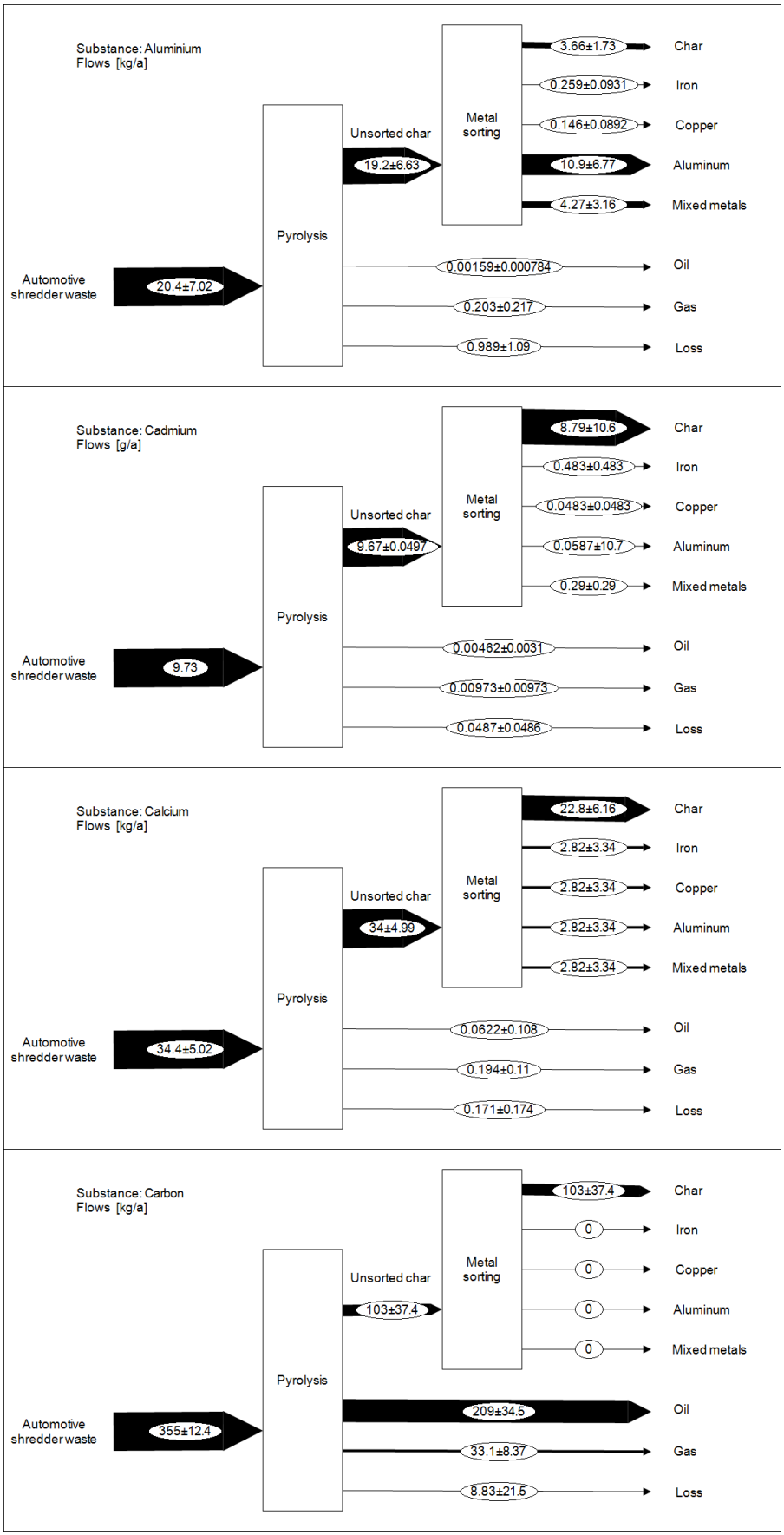


Figure 26 – Al, Cd, Ca, C substance flow analysis for pyrolysis of 1 ton of SR.

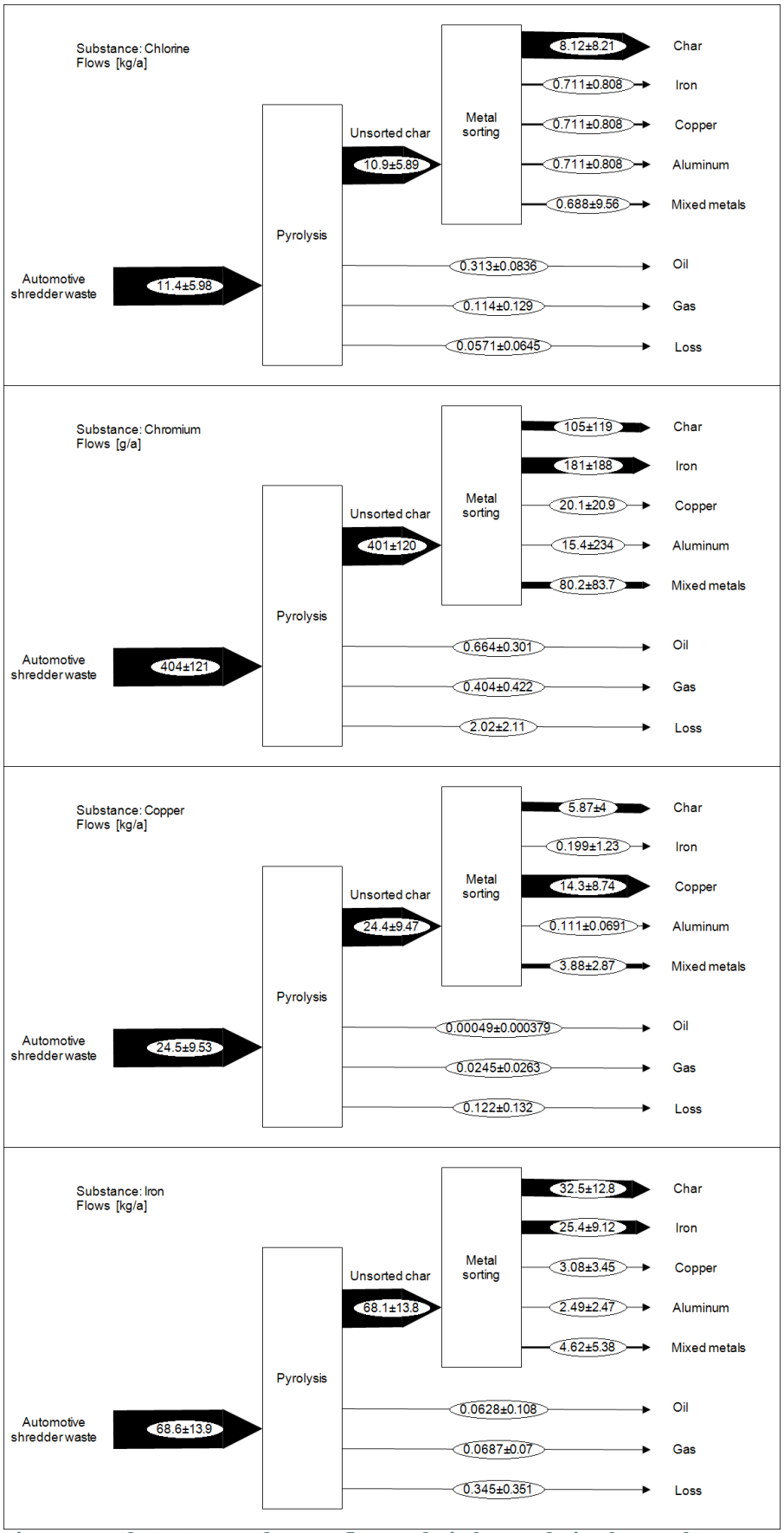


Figure 27 – Cl, Cr, Cu, Fe substance flow analysis for pyrolysis of 1 ton of SR.

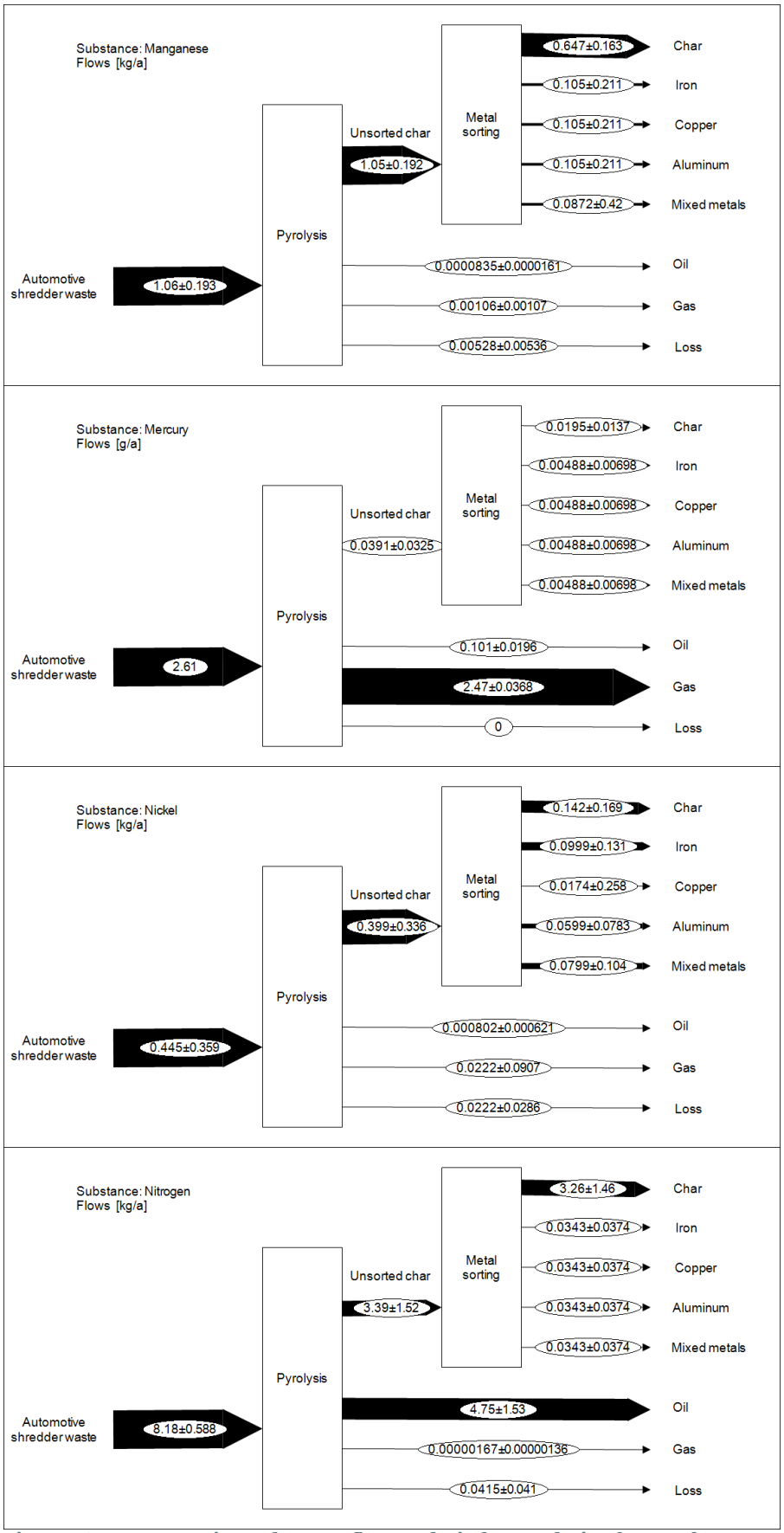


Figure 28 – Mn, Hg, Ni, N substance flow analysis for pyrolysis of 1 ton of SR.

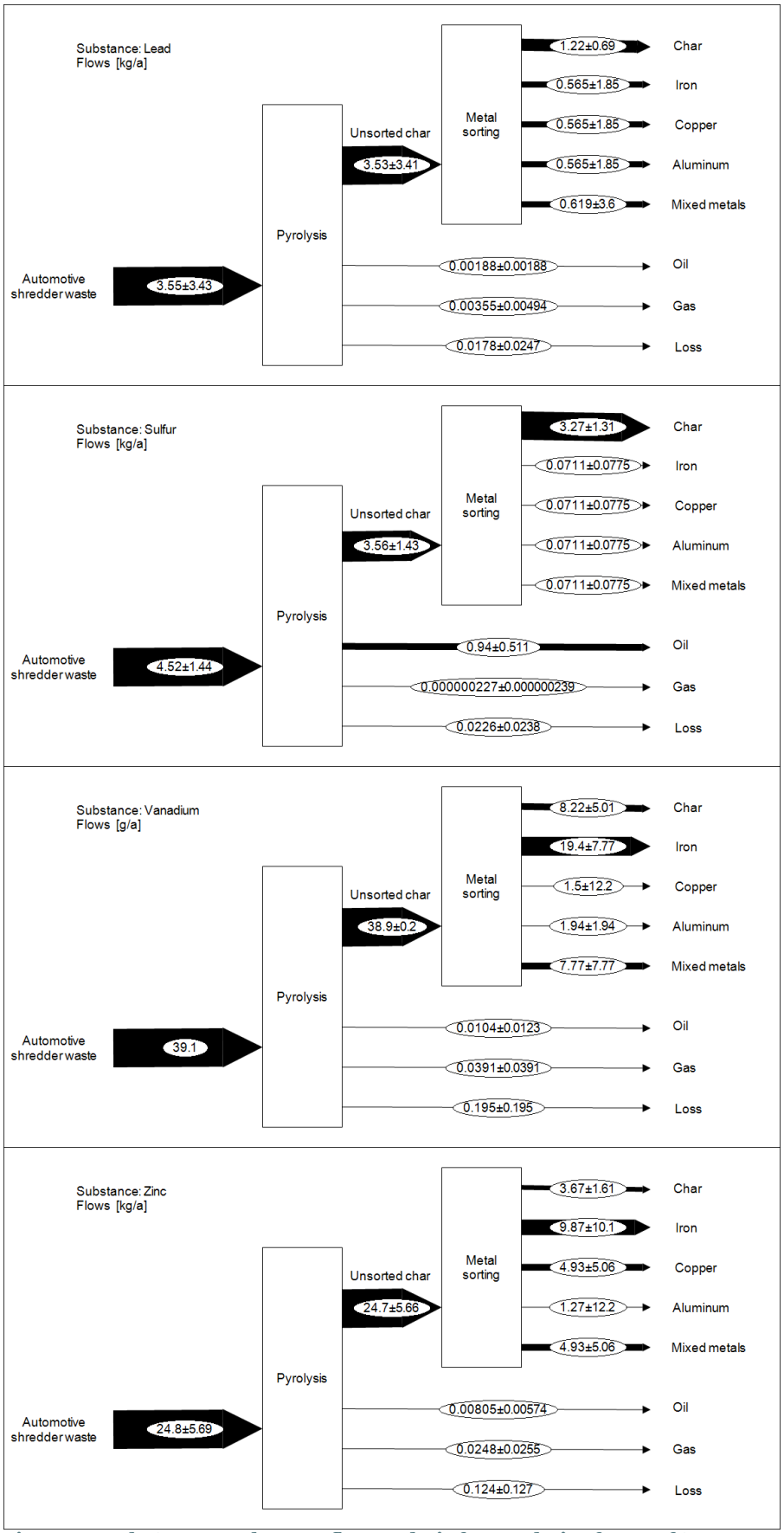


Figure 29 – Pb, S, V, Zn substance flow analysis for pyrolysis of 1 ton of SR.

Table 37 – LCI dataset of pyrolysis of SR.

Type	Parameter	Unit	mean	st.dev
Input	SR	ton	1	
	Electricity	MJ/ton ww	248.4	
	Pyrolysis oil	GJ/ton ww	0.29	
	Pyrolysis gas	GJ/ton ww	0.21	
Output	Char	% LHV, net	25.0	30.1
	Oil	% LHV, net	38.9	18.3
	Gas	% LHV, net	22.7	20.1
	Ferrous metals	kg/ton ww (% input)	25.9 (37.0)	9.31 (12.7)
	Non-Ferrous metals	kg/ton ww (% input)	25.7 (8.24)	11.3 (6.18)
	Unsorted metals	kg/ton ww (% input)	12.9 (6.73)	9.56 (7.70)
Transfer to char (% of content in waste input)	Al	% input	17.88	10.09
	C	% input	29.17	10.50
	Ca	% input	66.05	16.79
	Cd	% input	90.29	109.42
	Cl	% input	71.06	81.38
	Cr	% input	25.96	30.39
	Cu	% input	23.93	16.17
	Fe	% input	47.42	14.21
	Hg	% input	0.75	0.71
	Mn	% input	61.30	19.08
	N	% input	39.81	17.56
	Ni	% input	32.01	47.18
	Pb	% input	34.23	38.39
	S	% input	72.41	11.50
	V	% input	21.01	12.80
	Zn	% input	14.76	7.30
Transfer to oil (% of content in waste input)	Al	% input	0.008	0.005
	C	% input	58.999	9.510
	Ca	% input	0.181	0.313
	Cd	% input	0.048	0.032
	Cl	% input	2.738	1.609
	Cr	% input	0.164	0.089
	Cu	% input	0.002	0.002
	Fe	% input	0.091	0.158
	Hg	% input	3.877	0.749
	Mn	% input	0.008	0.002
	N	% input	58.009	18.281
	Ni	% input	0.180	0.202
	Pb	% input	0.053	0.073
	S	% input	20.80	12.02
	V	% input	0.027	0.031
	Zn	% input	0.032	0.024
Transfer to gas (% of content in waste input)	Al	% input	0.994	1.000
	C	% input	9.338	2.336
	Ca	% input	0.564	0.321
	Cd	% input	0.100	0.100
	Cl	% input	1.000	1.000
	Cr	% input	0.100	0.100
	Cu	% input	0.100	0.100
	Fe	% input	0.100	0.100
	Hg	% input	94.627	1.410
	Mn	% input	0.100	0.100
	N	% input	0.000	0.000
	Ni	% input	5.000	20.000
	Pb	% input	0.100	0.100
	S	% input	0.000	0.000
	V	% input	0.100	0.100
	Zn	% input	0.100	0.100

Appendix 5: LCI of disposal of SR in landfill

Gas emissions from landfill

Data regarding landfill gas production are taken from Møller et al. (2014) that, based on data regarding landfill gas production for landfilled SR provided by Scheutz et al. (2011), estimated an emission of CH_4 (including oxidation effect) in the order of 0.166 kg per kg of biogenic C over 100 years (i.e. the time horizon of the LCA). The amount of biogenic carbon still in the landfill after a 100 year period - calculated as the C input minus the amount of carbon lost as methane – is accounted as a CO_2 savings (Christensen et al., 2009).

Emission and production of other gases were also retrieved from Møller et al. (2014), based on data from Scheutz et al. (2010a) and Scheutz et al. (2010b), as shown in Table 38. The release of individual gases is the estimated considering that part of the produced gases are oxidised during their migration through the upper layer of the landfill.

Table 38 – Composition, production and release of gas from disposal of SR in landfill (from Møller et al, 2014).

	Composition	Oxidation	Production	Release after oxidation
	%v/v	%	kg/kg_{SR}	kg/kg_{SR}
CH4	27±0.8	29	$1.2*10^{-2}$	$8.4*10^{-3}$
CFC-11	2±1.2	90	$1.3*10^{-7}$	$1.3*10^{-8}$
HCFC-21	27±9.7	50	$1.8*10^{-6}$	$8.8*10^{-7}$
HCFC-31	20±7.8	30	$1.3*10^{-6}$	$9.1*10^{-7}$
HFC-41	1±0.3	0	$6.5*10^{-8}$	$6.5*10^{-8}$
CFC-12	1±0.5	30	$6.5*10^{-8}$	$4.6*10^{-8}$
HCFC-22	2±0.4	20	$1.3*10^{-7}$	$1.0*10^{-7}$
HFC-32	0±0.1	10	0.0	0.0
HFC-134a	10±5.1	0	$6.5*10^{-7}$	$6.5*10^{-7}$
HCFC-141b	2±0.5	20	$1.3*10^{-7}$	$1.0*10^{-7}$

Leachate production and composition

As described in Møller et al. (2014), data for modelling the production and composition of leachate produced from disposal of SR in a landfill is based on Hansen et al. (2011b), who performed leaching tests up to L/S of 9 l/kg. The amount of leachate produced in a 100 year period from disposal of SR in Danish landfill was estimated being 1.58 l/kg_{SR} (Møller et al., 2014). The release of individual compounds to the leachate phase was then extrapolated based by interpolating release data as a function of the L/S ratio (Møller et al. 2014).

As leachate generated in Danish landfills is typically treated before final discharge to surface (or marine) water bodies, the final release to the environment was then corrected to account for removal of individual compounds occurring during leachate treatment in wastewater treatment plant. An overview of inventory data on leaching during deposition of SR in Danish landfill and subsequent treatment of leachate at wastewater treatment plant is provided in Table 39. The uncertainty related to the composition of leachate is provided in Table 40, as estimated by Hansen et al. (2011b).

Table 39 – Leaching of individual substances during deposition of SR in Danish landfill and removal during leachate treatment in wastewater treatment plant (from Møller et al., 2014).

	kg/kg$_{SR}$	% of leachate content
Total-N	0,000452	9,80%
Total-P	5,48*10^{-6}	9,00%
Cl^-	0,000985	100,00%
F^-	4,02*10^{-6}	14%
SO_4^{2-}	2,18*10^{-5}	14%
HCO_3^-	0,007252	14%
NVOC (DOC)	0,00116	14%
$S - S^{2-}$	6,64*10^{-7}	14%
Al	1,08*10^{-7}	0,30%
As	1,03*10^{-7}	87,00%
Ba	1,76*10^{-6}	13,90%
Ca	0,000527	92,10%
Cd	7,14*10^{-11}	13,90%
Co	2,70*10^{-9}	17,00%
Cr (VI)	2,33*10^{-9}	11,20%
Cr (tot)	2,02*10^{-8}	11,20%
Cu	6,98*10^{-9}	2,00%
Fe	4,14*10^{-5}	1,70%
Hg	1,73*10^{-10}	10,30%
K	0,000451	94,30%
Mg	0,000704	95,10%
Mn	1,34*10^{-6}	47,10%
Mo	9,58*10^{-9}	95,00%
Na	0,000854	89,20%
Ni	1,16*10^{-7}	47,10%
Pb	7,14*10^{-10}	2,70%
Sb	6,66*10^{-9}	45,80%
Se	2,33*10^{-9}	14%
Si	4,28*10^{-5}	14%
V	3,14*10^{-8}	16,90%
Zn	9,60*10^{-8}	7,30%
Benzene (sum)	-3*10^{-7}	30%
PAH	-6,00*10^{-11}	29%
PAH	-6,00*10^{-11}	29%

Table 40 – Estimated uncertainty of characteristic of leachate from SR disposed in landfill (taken from Hansen et al., 2011a).

Parameter	Data variation (%)
Cl^-	60
F^-	38
SO_4^{2-}	39
Al	48
As	16
Ba	29
Ca	39
Cd	74
Co	37
Cr (total)	68
Cu	23
Fe	93
Hg	131
K	19
Mg	26
Mn	34
Mo	31
Na	44
Ni	47
Pb	71
Sb	34
Se	2
Si	45
V	68
Zn	61
DOC/NVOC	37

Appendix 6: LCI of co-combustion of SR in a cement kiln

Data regarding co-combustion of SR in a cement kiln are not available in Denmark, as this option is currently non-existing. The inventory dataset has thus been developed based on literature information. In particular, Vermeulen et al. (2012) reviewed existing studies and provided for individual compounds ranges of emissions to air occurring from co-combustion of SR in a cement kiln. With regards to solid residues, it was considered that the inert part of SR (i.e. ash and metals) would end up in the clinker and thus no further modelling was included.

Data on SR composition and emissions to air provided by Vermeulen et al. (2012) are reported in Table 41, together with the waste composition used. The TCs to air (TC_{air}) were calculated according as:

$$TC_{air} = \frac{Output_{air}}{Input_{waste}}$$

where $Input_{waste}$ is the content of a specific compound in the waste and $Output_{air}$ is the emission of such compound to the atmosphere. Using a uniform distribution for $Output_{air}$ (minimum and maximum showed in Table 41) and a lognormal distribution for $Input_{waste}$ (assuming average as the geometrical mean and mix/max as the 2.5/97.5% range Table 41), the TC_{air} where computed 15000. The obtained results were characterized by a log-normal distribution, for which median and variance were calculated, as shown in Tabel 42. The fact that the TC for C is very close to 100% in Table 42 indicates that the calculation approach is correct, as all organic C contained in SR should be converted to CO2 during a complete combustion of the feedstock. However, as a TC >100% does not make sense from a physical point of view, it was decided to adopt a TC to air for C equal to 100% for the inventory modelling. The number of simulations was decided based on the following formula:

$$\varepsilon = \frac{3\sigma}{\sqrt{N}} \rightarrow N = \left(\frac{3 \times \sigma}{\varepsilon}\right)^2 = \left(\frac{3 \times stdevp(min, max, average(min, max))}{50}\right)^2$$

where ε is the error of the estimation (here assumed maximum 2%), σ is the deviation of the data and N the number of simulations. The value N was estimated for all individual compounds using data in Table 41, and a rounding of the maximum value (i.e. 14902) being then selected for running the simulation.

Table 41 – Data regarding emissions to air and waste composition, as provided by Vermeulen et al. (2012).

Compound	Unit	Air emission		Waste composition		
		min	max	min	max	average
C	kg/ton_{SR}	1.504		279	626	410
N	kg/ton_{SR}			8.8	45	19
S	kg/ton_{SR}	0.0990	0.2295	1.9	5.6	3.7
As	g/ton_{SR}	2.97E-03	6.83E-03	1.2	70	30
Cd	g/ton_{SR}	3.42E-03	1.45E+00	2	86	34
Cr	g/ton_{SR}	1.40E-02	2.81E-02	17	7000	1120
Cu	g/ton_{SR}	4.57E-01	1.97E+00	27	16,600	4910
Hg	g/ton_{SR}	1.03E+00	2.02E+00	0.2	14	4.1
Ni	g/ton_{SR}	7.34E-03	2.20E-01	54	4000	734
Pb	g/ton_{SR}	3.92E-01	5.48E+00	94	7000	2610
Zn	g/ton_{SR}	6.61E-01	3.61E+01	1430	14,100	8260

Table 42 – TCs (%) to air for co-combustion of SR in a cement kiln.

	TCs (%) to air	
Compound	**median**	**deviation**
C	102.0	257.8
S	0.0397	0.1071
As	0.0001	0.0003
Cd	0.0172	0.0478
Cr	0.0005	0.0013
Cu	0.0273	0.0790
Hg	0.0367	0.1038
Ni	0.0022	0.0089
Pb	0.0571	0.2195
Zn	0.3730	1.2975

Appendix 7: Inventory datasets

Inventory dataset for plastic recycling

Inventory data for recycling of plastic are provided in Table 43 and Table 44, together with the related datasets used in the LCI modelling. The dataset is based on information retrieved from Franklin Associates (2011).

Table 43 – Inventory of input of material and energy to the plastic recycling process.

Type	Process	Unit	Amount
Water	Water from Waterworks, Sweden, 2008	kg/kg_{ww}	0.02
Diesel	Production and Combustion of Diesel Oil in Truck, EU2, 1998	kg/kg_{ww}	$1.8e^{-4}$
Virgin HDPE*	Polyethylene, HDPE, granulate, at plant, RER	kg/kg_{ww}	-0.93
Electricity	Marginal Electricity Consumption incl. Fuel Production, Coal, Energy Quality, DK, kWh, 2006	kWh/kg_{ww}	0.49
Natural gas	Natural Gas (prod + comb), <1 MW, kg, Denmark, 1990	kg/kg_{ww}	$7.7e^{-3}$
NaOH	Sodium hydroxide (NaOH), RER, ELCD, 1996 - corrected	kg/kg_{ww}	0.025
* Avoided production			

Table 44 – Inventory of emissions from plastic recycling process.

Name	Compartment	Sub compartment	Unit	Amount
BOD_5	water	surface water	kg/kg_{ww}	1.9/1000
Cd^+	water	surface water	kg/kg_{ww}	$6.1e^{-10}$
CO_2, fossil	air	unspecified	kg/kg_{ww}	0.0036
COD	water	surface water	kg/kg_{ww}	0.021
N	water	surface water	kg/kg_{ww}	$3e^{-6}$
P	water	surface water	kg/kg_{ww}	$1e^{-6}$
SO_2	air	unspecified	kg/kg_{ww}	$1.1e^{-6}$
TOC	water	surface water	kg/kg_{ww}	0.0064

Inventory dataset for electricity and heat production

Electricity

Life Cycle Inventory and documentation for 1 kWh marginal Danish electricity "Marginal Electricity Consumption incl. Fuel Production, Coal, Energy Quality, DK, kWh, 2006" from EASETECH database.

Technology

The production of 1 kWh of electricity is calculated as an average of seven Danish power stations (six cogeneration plants and one plant that produces only electricity). These plants were identified as those plants that can regulate the electricity production according to market demand, i.e. they are the "marginal" power stations.

Input

Coal (primary fuel) and oil.

Output

1 kWh of electricity supplied to consumers. The LCI includes a loss of 2% in distribution.

Process

Coal and oil are used for energy production. Air emissions are included as well as solid waste fractions from the combustion process. Mining, processing and transportation of coal and oil are included in the LCI.

Table 45 – Inventory dataset "electricity "Marginal Electricity Consumption incl. Fuel Production, Coal, Energy Quality, DK, kWh, 2006".

Parameter	Compartment	Sub-compartment	Unit	Amount
As	Air	Unspecified	kg	$4.6e^{-09}$
Cd	Air	Unspecified	kg	$3.5e^{-10}$
CO_2, fossil	Air	Unspecified	kg	0.92
CO, fossil	Air	Unspecified	kg	$9.9e^{-05}$
Cr	Air	Unspecified	kg	$6.6e^{-09}$
Cu	Air	Unspecified	kg	$4.4e^{-09}$
Pb	Air	Unspecified	kg	$6.7e^{-09}$
Hg	Air	Unspecified	kg	$9.2e^{-09}$
Ni	Air	Unspecified	kg	$9,5e^{-09}$
NOx	Air	Unspecified	kg	0,000661
N_2O	Air	Unspecified	kg	$8.1e^{-06}$
NMVOC	Air	Unspecified	kg	$1.3e^{-05}$
Se	Air	Unspecified	kg	$6.3e^{-08}$
SO_2	Air	Unspecified	kg	$2.2e^{-04}$
Zn	Air	Unspecified	kg	$1.4e^{-08}$
CH_4, fossil	Air	Unspecified	kg	$1.3e^{-05}$
PM, < 2.5 um	Air	Unspecified	kg	$3.5e^{-05}$
Input				
Heavy fuel oil, EU-15, ELCD, 2003- corrected			kg	0.00493
Hard coal, EU-27, ELCD, 2002 - corrected			kg	0.394

Location
Denmark
Year
2006

Data Sources
The data originate from a survey conducted by Energinet.dk and Dansk Standard, aiming at producing environmental declaration of the Danish electricity which is influenced by market demand, thus not for an average kWh. Data has since been updated from 2005 to 2007. Emissions of heavy metals are averages from three Danish CHP plants (Green Account 2006 for Esbjerg [2] , Asnæs (line 2 +5) [3] and Stigsnæs (line 1 +2) [4]).

Data Quality (dqi = good)
Data were collected as an average of seven Danish installations but adjusted so that coal was the dominant fuel. This is because coal is the fuel that will be affected when the plant responds to market demand [1]. Heavy metal emissions are based on three power plants and are considered to be relatively well documented.

Note
Electricity is produced at six of the seven plants together with heat. Therefore, resource consumption, emissions, etc. are allocated between the two products. This is done in [1] based on energy quality, which is a method that allocates a relatively large proportion of resource consumption and emissions for electricity production. Emissions of heavy metals were allocated in a similar way.

Referencer
1. Behnke, K. (2006): Notat om deklaration af fremtidigt elforbrug, Energinet.dk, Denmark.
2. DONG Energy (2007): Esbjergværket. Grønt regnskab 2006.
3. DONG Energy (2007): Asnæsværket. Grønt regnskab 2006.
4. DONG Energy (2007): Stigsnæsværket. Grønt regnskab 2006

Heat

Life Cycle Inventory and documentation for 1 kWh marginal Danish heat "District Heating, marginal average, (DK), kWh, 2012" from EASETECH database.

Table 46 - Inventory dataset "District Heating, marginal average, (DK), kWh, 2012"

Emission	Compartment	Sub-compartment	Unit	Amount
.CH4, fossil	Air	Unspecified	kg	0,000175
CO2, fossil	Air	Unspecified	kg	0,15
N2O	Air	Unspecified	kg	5,76E-06
PM	Air	Unspecified	kg	0,000283
NOx	Air	Unspecified	kg	0,000407

The LCI was prepared by Energistyrelsen (2011): "Forudsætninger for samfundsøkonomiske analyser i energisektoren", as described in Jakobsen et al. (2013), page 79-80. Fuel composition was 22% wood, 21% natural gas, 20% waste, 16% coal, 8% straw, 7% oil and 5% biogas.

Appendix 8: Evaluation of the potential environmental impacts

Characterized potential environmental impacts

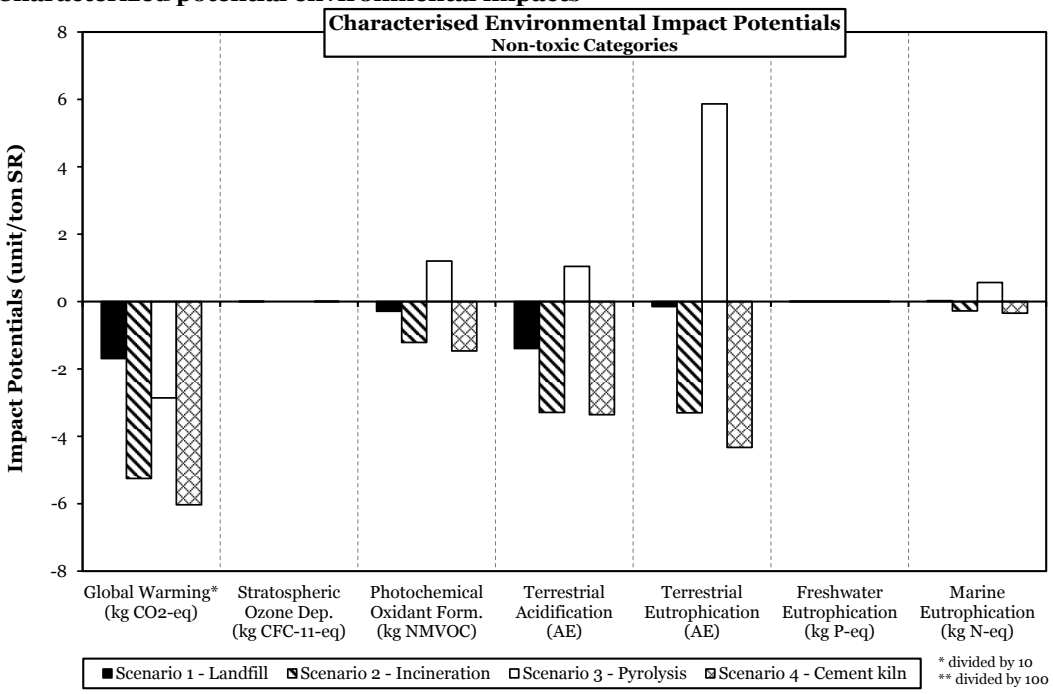


Figure 30 - Characterised potential non-toxic impacts from the treatment of 1 ton of SR in the four analysed scenarios (PE = person equivalent).

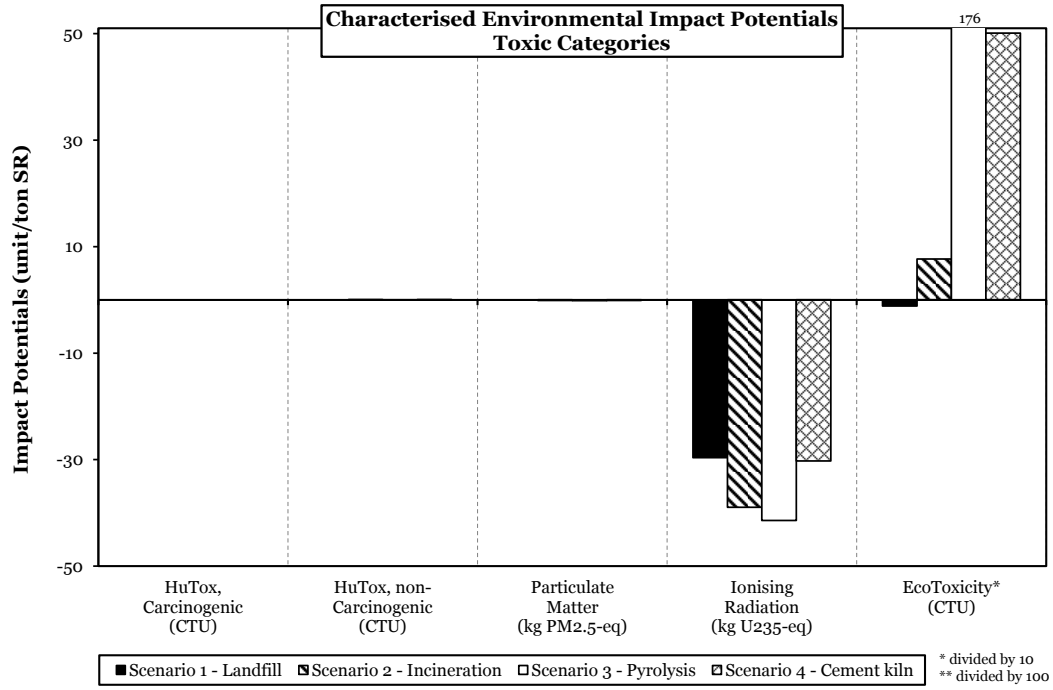


Figure 31 - Characterised potential toxic impacts from the treatment of 1 ton of SR in the four analysed scenarios (PE = person equivalent).

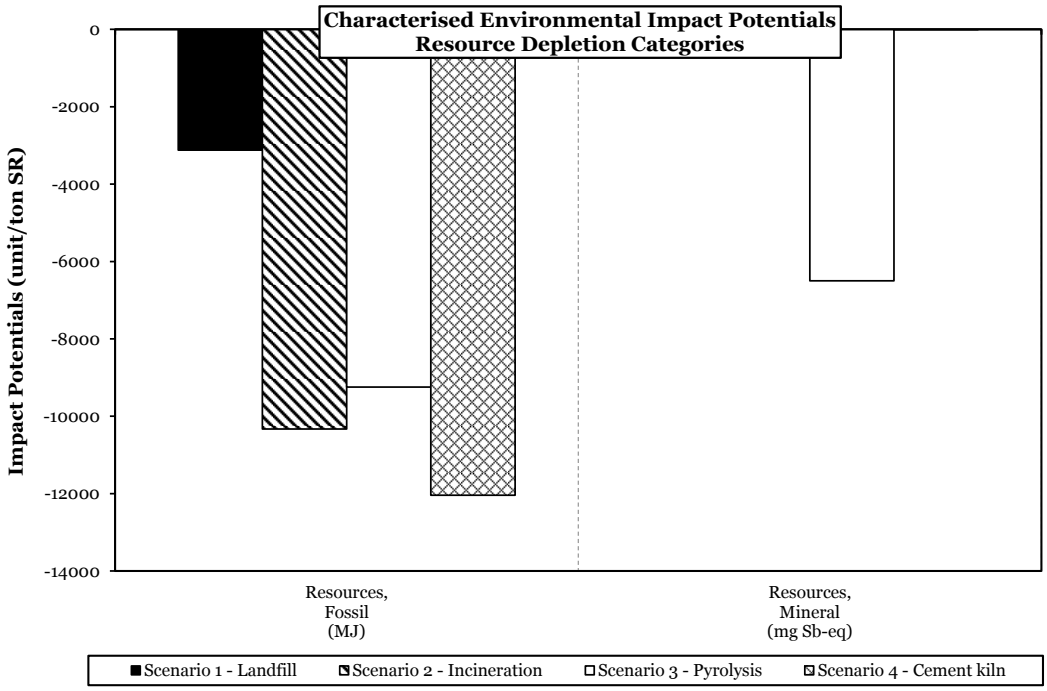


Figure 32 - Characterised potential Resource Depletion from the treatment of 1 ton of SR in the four analysed scenarios (PE = person equivalent).

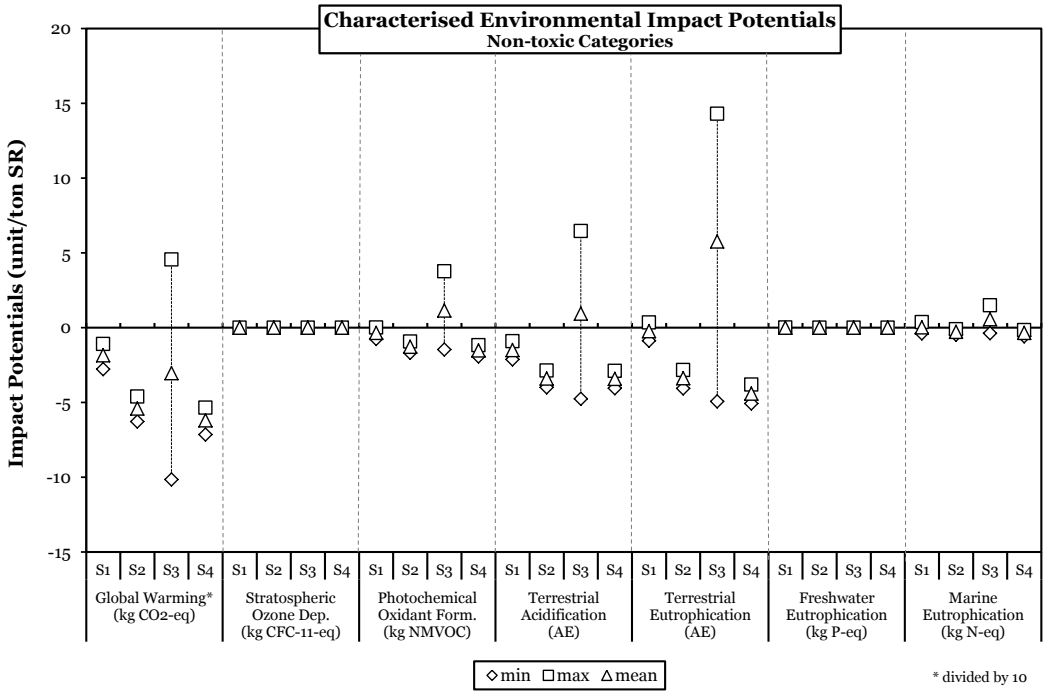


Figure 33 – Characterised potential non-toxic impacts from the treatment of 1 ton of SR in the four analysed scenarios (PE = person equivalent), including min-max range.

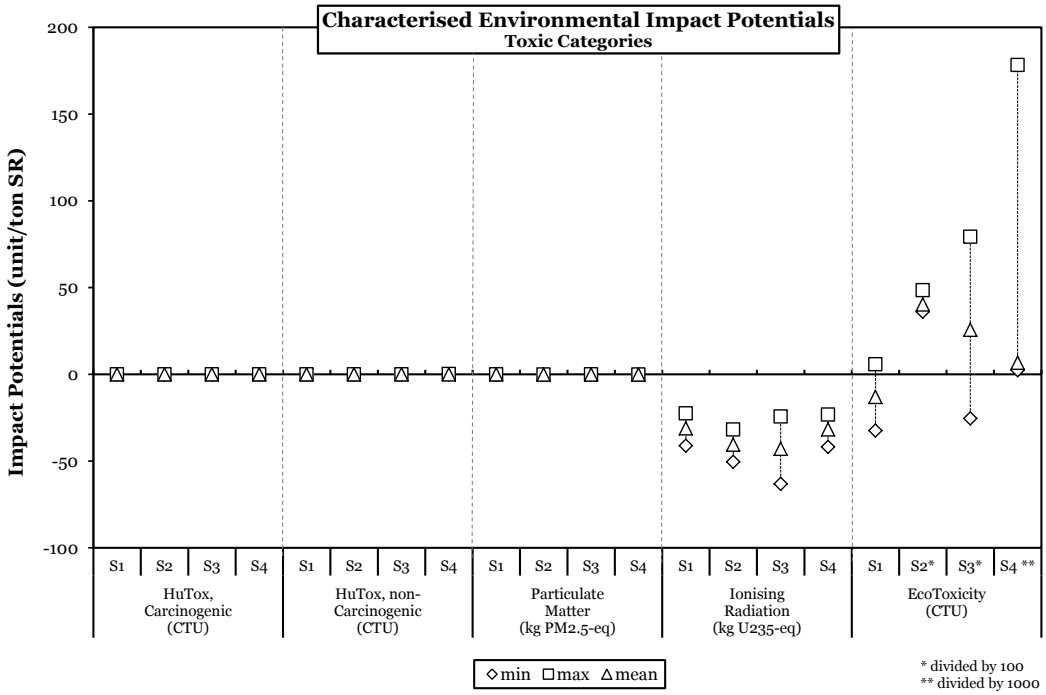


Figure 34 - Characterised potential toxic impacts from the treatment of 1 ton of SR in the four analysed scenarios (PE = person equivalent), including min-max range.

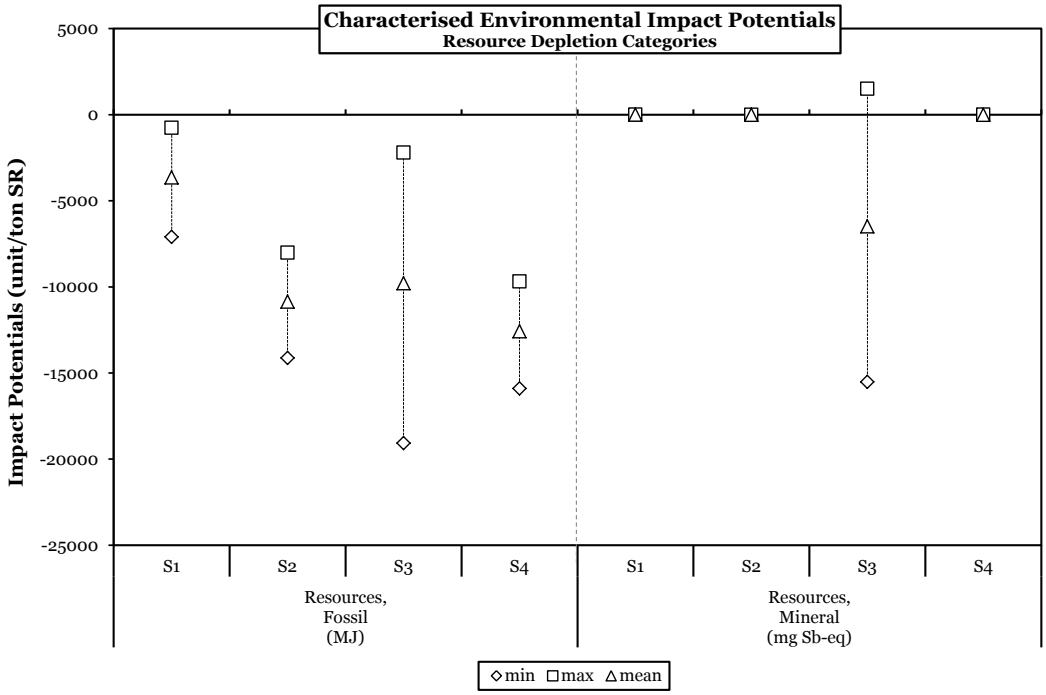


Figure 35 - Characterised potential Resource Depletion from the treatment of 1 ton of SR in the four analysed scenarios (PE = person equivalent), including min-max range.

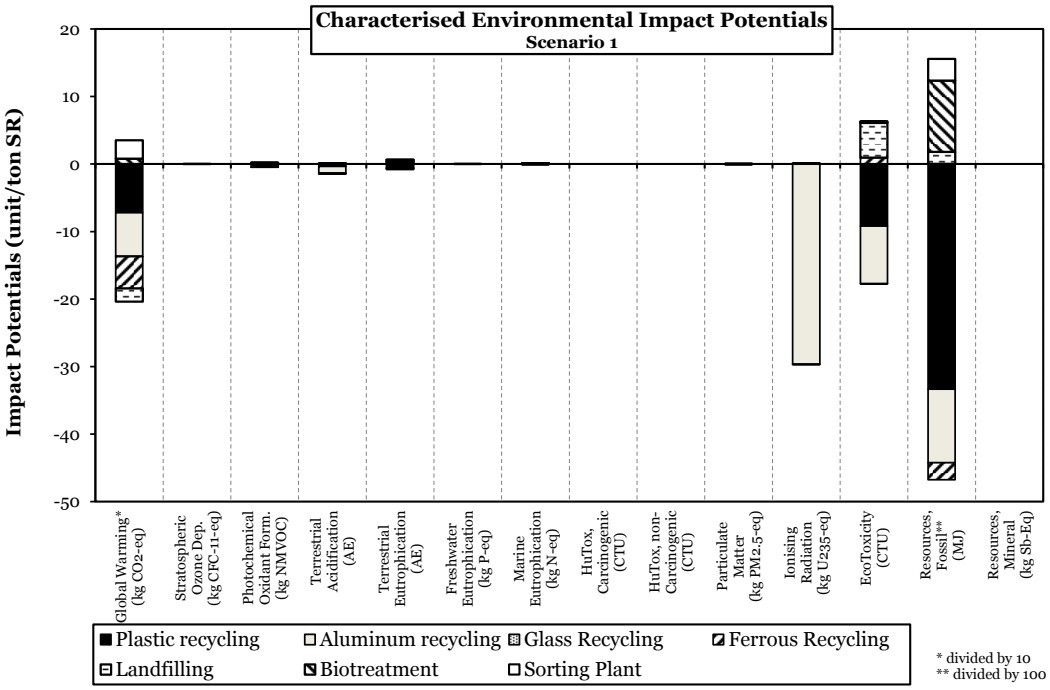


Figure 36 - Characterised potential impacts from the treatment of 1 ton of SR in Scenario 1, disaggregated according to individual treatment processes (PE = person equivalent).

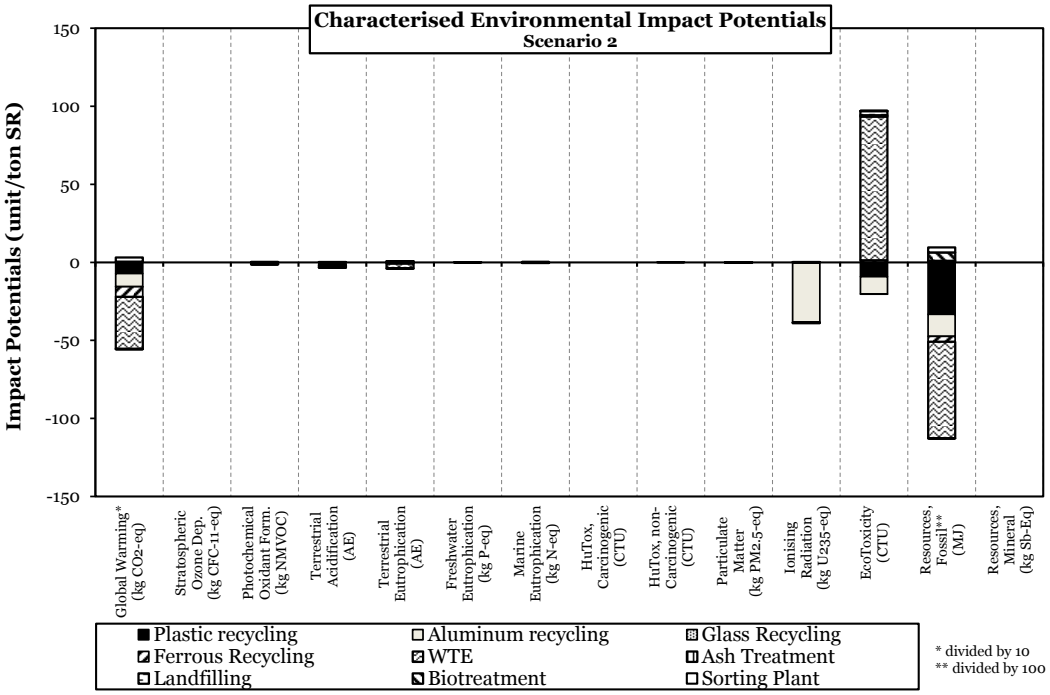


Figure 37 - Characterised potential toxic impacts from the treatment of 1 ton of SR in Scenario 2, disaggregated according to individual treatment processes (PE = person equivalent).

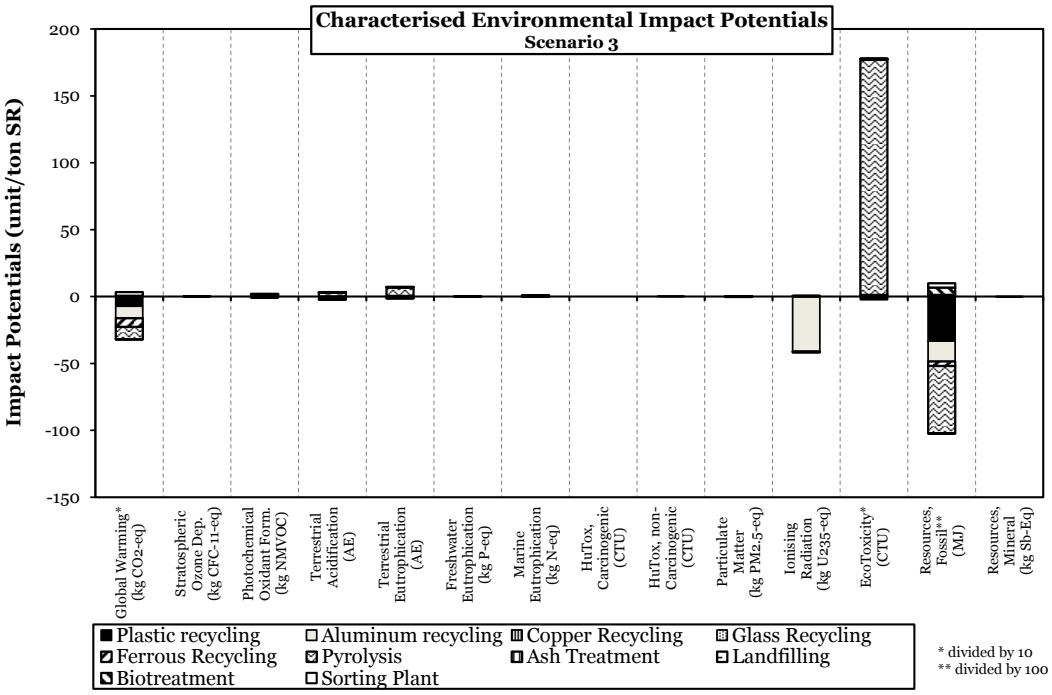


Figure 38 - Characterised potential toxic impacts from treatment of 1 ton SR in Scenario 3, disaggregated according to individual treatment processes (PE = person equivalent).

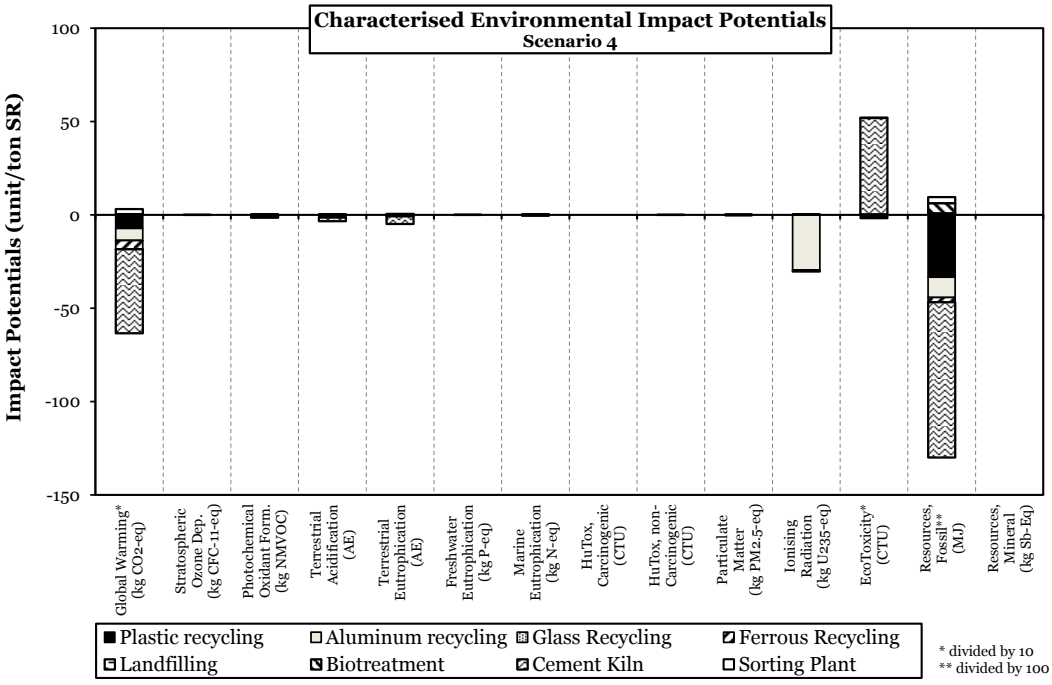


Figure 39 - Characterised potential toxic impacts from the treatment of 1 ton of SR in Scenario 4, disaggregated according to individual treatment processes (PE = person equivalent).

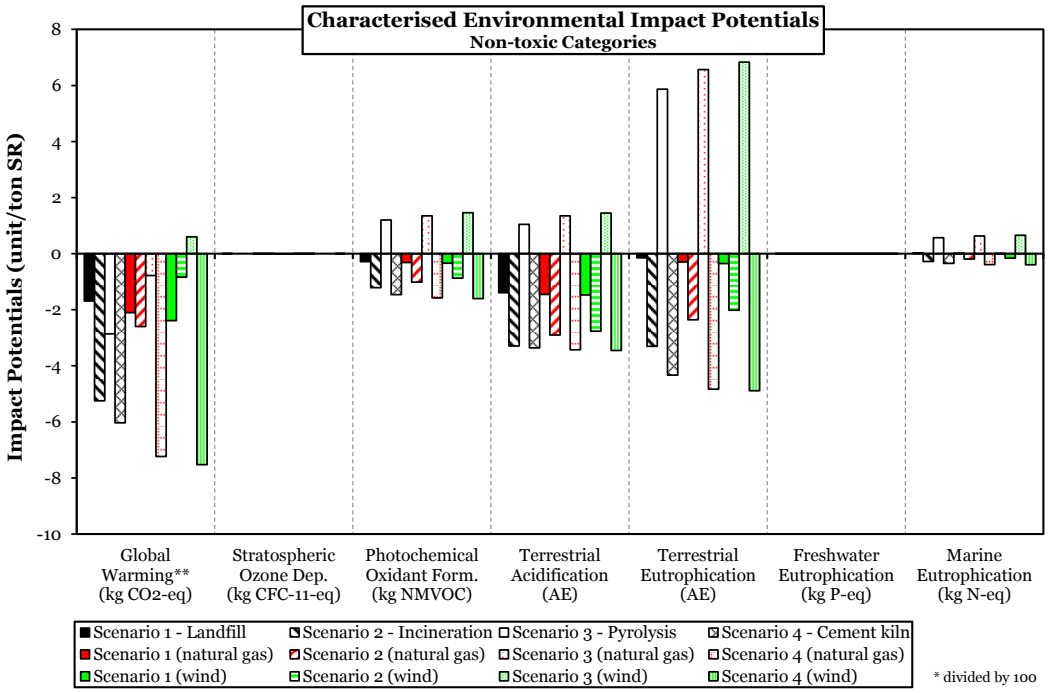


Figure 40 - Characterised potential non-toxic impacts from the treatment of 1 ton of SR in the four analysed scenarios modelled using alternative marginal electricity production (i.e. coal, natural gas and wind). (PE = person equivalent).

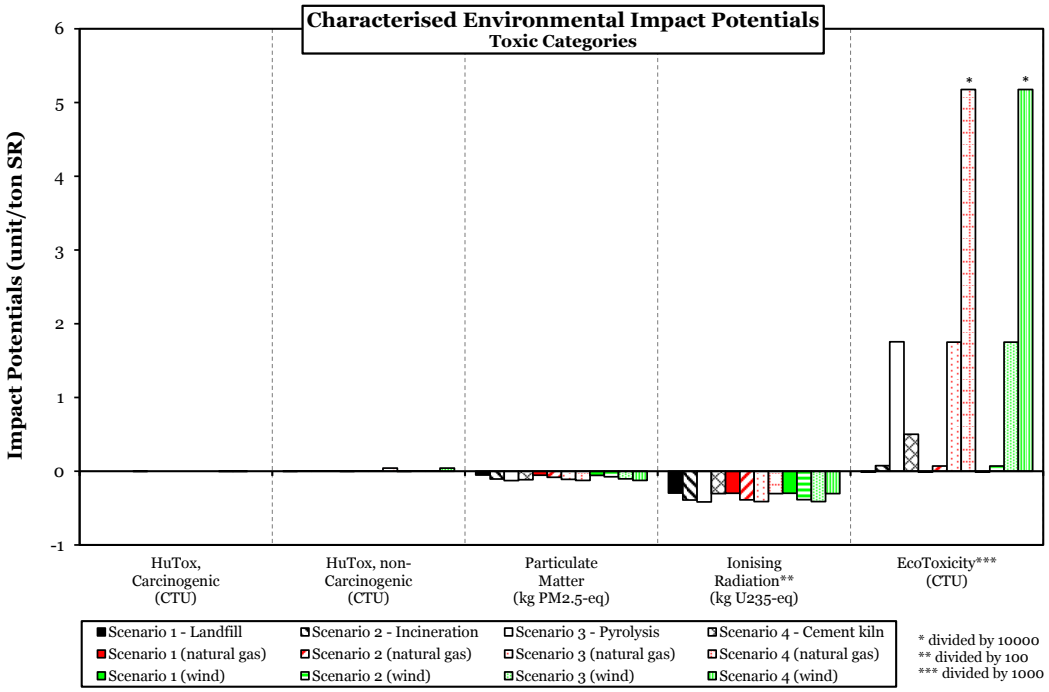


Figure 41 - Characterised potential toxic impacts from the treatment of 1 ton of SR in the four analysed scenarios modelled using alternative marginal electricity production (i.e. coal, natural gas and wind). (PE = person equivalent).

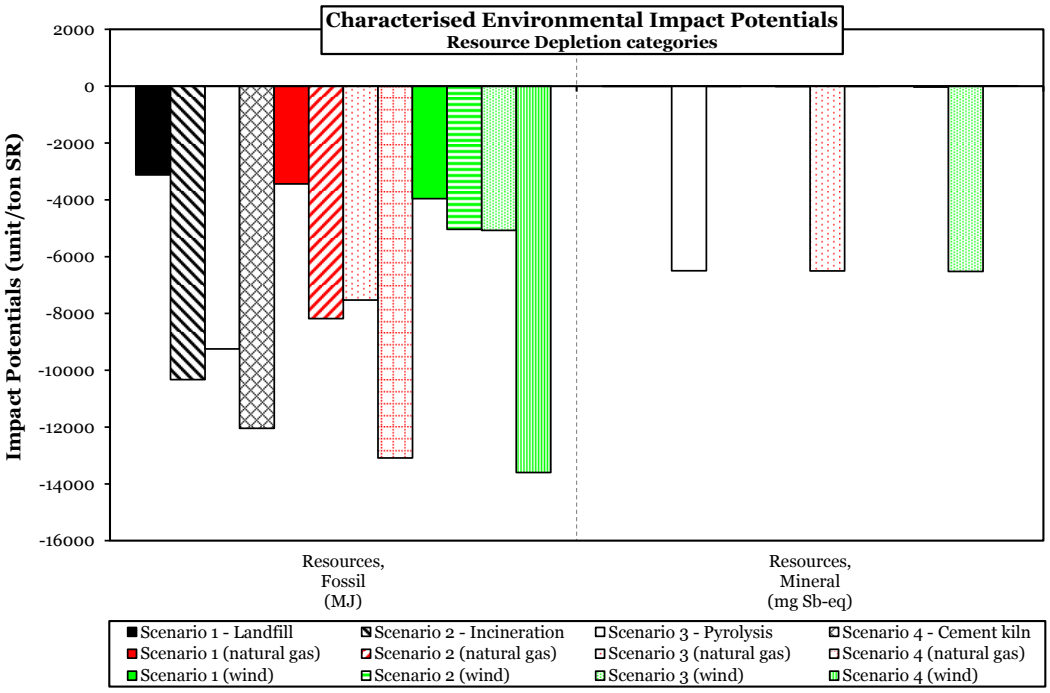


Figure 42 – Characterised potential Resource Depletion from the treatment of 1 ton of SR in the four analysed scenarios modelled using alternative marginal electricity production (i.e. coal, natural gas and wind). (PE = person equivalent).

Potential environmental impacts by material fraction

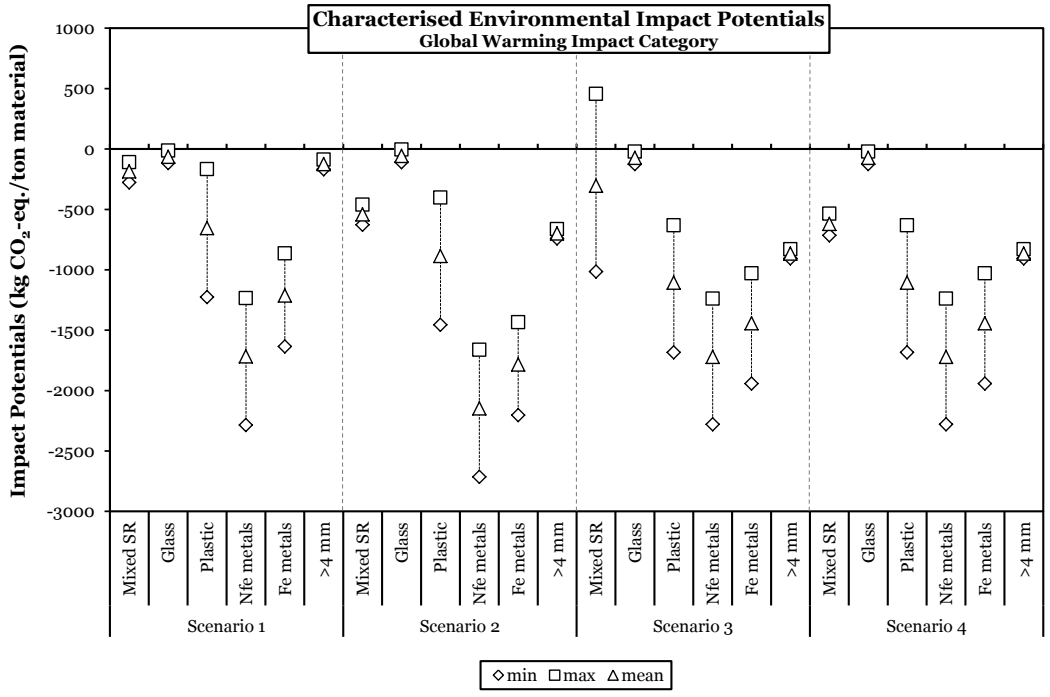


Figure 43 - Characterised potential Global Warming impacts from treatment of 1 ton SR in the four analysed scenarios, disaggregated according to individual material fractions (PE = person equivalent).

Appendix 9: Review report from Teknologisk Institut

TEKNOLOGISK INSTITUT

Critical review of life cycle assessment of shredder residue management

Review performed by:
Bjørn Malmgren-Hansen, Mathias Sehested Høeg Kemner and Trine Henriksen
Danish Technological Institute
July 2014

1 Scope

Method of review

The critical review has been performed following the guidelines for critical review of life cycle studies described in:

- ISO 14040 (Environmental Management - Life Cycle Assessment - Principles And Framework)
- ISO 14044 (Environmental Management - Life Cycle Assessment - Requirements And Guidelines).

It has been checked that this LCA is consistent with these standards, if the methods used are scientifically and technically valid, if the applied data are suitable according to the goal of the study and whether the report is transparent and consistent and reflects the limitations and goal of the study.

The reviewers

The critical review has been performed by The Danish Technological Institute (DTI). Trine Henriksen and Mathias Sehested Høeg Kemner works with life cycle assessment in both government funded R&D projects as well as with private companies. Bjørn Malmgren-Hansen has large insight in the technical aspects of shredder waste management processes.

Scope of review

The EASETECH model, which has been built for calculating the LCI and LCIA results in this project, has not been part of the review. The LCI and technology description have been evaluated by evaluation of the descriptions in the report by DTU Environment.

Prior to the draft version of the LCA-report, which is the basis of this review, a preliminary draft has been commented by DTI. These comments were mainly on the structure of the report and its formulations.

2 Review process

Review of the preliminary draft version of the LCA-report was performed in June 2014.

In the final draft version the comments from the preliminary review has been well incorporated by DTU Environment.

Review of the final draft version and the writing of this review report were performed in July 2014.

This report sums up the conclusions on the LCA's consistency with the ISO standards and best practice.

3 Review

The report

The LCA report has a well-built and logical structure, which gives a natural flow of information in order to understand the background and the results of the LCA study. However, the summary needs to include a description of the evaluated scenarios, in order to improve the readability and the understanding of the presented results.

The report follows the overall reporting guidelines from ISO 14044, with sections for goal and scope, inventory analysis, impact assessment and interpretation of results. Furthermore, the report is transparent in terms of the methods used.

Methods

The methods used in the LCA study are considered to be consistent with the ISO standards on life cycle assessment. The goal and scope includes all necessary information on system boundaries, data quality, limitations and timeframe as well as a well-defined functional unit.

The LCA study is based on a consequential approach in terms of modeling multi output processes, thus dealing with allocation issues by expanding the system boundaries to include avoided burdens. This is considered to be in line with the goal of the study, which is to evaluate possible changes in the treatment system for shredder residues and the subsequent environmental impacts.

Data

The report gives generally a good insight into the data used. This is supplemented with quality indicator values for the datasets used in the model, which in all gives a very good understanding of the data and its quality.

The report gives adequate background information on inventory data sources.

Technology

The report has good descriptions/overviews of the treatment processes with exhaustive explanations of the different technologies and process steps.

Especially the conclusions about pyrolysis are considered reasonably. (That implementation of this kind of treatment should be supported by pilot- or full-scale testing).

Results

The life cycle impact assessment (LCIA) results are presented as normalized potentials, both with and without a minimum-maximum range and mean-value. To present the indicator results with this indication of the uncertainty makes good sense, in order to graphically depict the great uncertainty in the pyrolysis datasets relative to the other technologies, hence conclude that the pyrolysis technology should undergo further testing before being implemented.

In addition to the above, the results are further divided into process impacts and material fractions impacts in the climate change impact category. This gives a very detailed understanding of the environmental performance of the systems under study, and enables the reader to understand how single processes and fractions contribute to the overall scenario results.

The discussion and interpretation of the results of the impact assessment has a level of detail corresponding to the graphical representation of the results, thus being sufficient in order to highlight the most relevant conclusions.

It should however be mentioned that only the normalized results are shown, i.e. results given in person-equivalents. According to the ILCD Handbook normalization is an optional step in LCAs and the LCIA results should be shown as individual impact categories with each their own indicator.

4 Conclusions

The LCA study that has been reviewed is considered to be consistent with the requirements on LCA studies described in ISO 14044.

The report documenting the LCA study is very well structured and has a high level of detail in terms of technology description, inventory data and presentation and interpretation of results.

Life cycle assessment of shredder residue management

The present report includes a life cycle assessment (LCA) of the treatment of shredder residues in Denmark. The assessment compares the potential environmental impacts and depletion of abiotic resources in relation to four alternative scenarios, all including the sorting of recyclables and the management of residual material through thermal treatment and/or disposal in controlled landfills after biological stabilisation.

The following can be concluded:

- Diverting SR >4 mm from landfill provides benefits from an environmental perspective and should thus be supported.
- Sorting and recycling metals, plastics and glass are beneficial to the environment and should thus be continued. Special focus should be given to increasing metal recovery, as this provides the greatest environmental benefits.
- Incineration seems currently the best option for the treatment of >4 mm residues.
- Pyrolysis seems to have worse energy efficiency but better downstream metal recovery (i.e. from the residues) than incineration. However, a clear conclusion could not be drawn, because results for pyrolysis were associated with significant uncertainty, owing to the lack of precise inventory data describing the process. Thus, any decision regarding the implementation of pyrolysis for treating >4 mm SR fractions should first be supported by pilot- to full-scale tests of the process, to deliver a better understanding thereof.

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